

II. SPECIES ACCOUNTS

A. Focal Listed Species

a. *Cirsium hydrophilum* var. *hydrophilum* (Suisun thistle)

1) *Brief Overview*

Cirsium hydrophilum var. *hydrophilum* was listed as endangered in its entire range on November 20, 1997 (U.S. Fish and Wildlife Service 1997a). It has a recovery priority number of 3C, based on a high degree of threat, a high potential of recovery, and its taxonomic standing as a subspecies. The “C” ranking indicates some degree of conflict between the conservation needs of the species and economic development (U.S. Fish and Wildlife Service 1983). It is not listed as endangered or threatened by the State of California. Habitat loss is the primary cause of decline in this species. Currently, two or three *populations*, are thought to be extant.

2) *Description and Taxonomy*

Description. *Cirsium hydrophilum* (E. Greene) Jepson var. *hydrophilum* (Suisun thistle) is a *perennial* herb in the Asteraceae (aster) family (**Figure II-1**). In the pre-flowering phase it grows as a short, broad, vegetative *rosette* with large leaves, approximately 0.3 to 0.9 meter (1 to 3 feet) long. The leaves have deep lobes with spines up to 1 centimeter (nearly 0.5 inch) long at the tips. The upper leaf surface of the youngest basal leaves are covered with hairs, but typically become smooth and somewhat glossy with maturity. In contrast, the lower leaf surface retains a thick white covering of hairs even when mature. The juvenile vegetative phase lasts until plants are large enough to flower (Keil and Turner 1993). During the mature phase the *rosette* bolts, and develops a tall (1 to 1.5 meters [3 to 5 feet]) leafy stem in the second year or later. Stems are typically branched above the middle of the main stem, but up to 15 stems may occasionally branch from the base of single large plants (P. Baye unpubl. data 2000). Leaves on stems are much smaller, more deeply lobed, and spinier than juvenile leaves of the *rosette*. The reduced stem leaves either clasp the stem at their bases, or have ear-like appendages near the stem. Stem leaves become progressively smaller near the top of the plant. The egg-shaped flowerheads (2.5 centimeters [1 inch] long) are composed of small individual *florets* united into a single unit. Many rose-purple *corollas* protrude. Flowerheads occur either as solitary units or in clusters. The bracts of the flowerheads have a distinct green, glutinous ridge on the back that distinguishes *Cirsium hydrophilum* var. *hydrophilum* from other *Cirsium* species in the area. The cypsalae, seed-like dry fruits similar to an achene, are about 4 to 5 millimeters (0.2 inch) long, and glossy dark brown to black with thick, hard outer walls (Munz 1959, Keil and Turner 1993, P. Baye unpubl. data 1999-2000).



FIGURE II-1. *Cirsium hydrophilum* var. *hydrophilum* (Photo Credit Valary Bloom, USFWS)

Taxonomy. *Cirsium hydrophilum* var. *hydrophilum* was originally described as *Cnicus breweri* Gray var. *vaseyi* Gray (Gray 1888). *Cnicus breweri* is a *taxon* now placed in *Cirsium douglasii* DC var. *breweri* (A. Gray) (Keil and Turner 1993). Subsequent synonyms, now invalid, include *Carduus hydrophilus* Greene (Greene 1892) and *Cirsium vaseyi* (Gray) Jepson var. *hydrophilum* (Greene) Jepson (Jepson 1925). Jepson (1901) was the first to apply the combination *Cirsium hydrophilum*. The species *Cirsium hydrophilum*, as now interpreted, (Howell 1969, Keil and Turner 1993) comprises two morphologically similar varieties: *Cirsium hydrophilum* var. *vaseyi*, (synonym: *Cirsium vaseyi* [A. Gray] Jepson), a related rare thistle *endemic* to seeps in serpentine soils on Mount Tamalpais, Marin County, and *Cirsium hydrophilum* var. *hydrophilum*, *endemic* to *brackish tidal* marshes in Suisun Marsh, Solano County.

The two varieties of *C. hydrophilum* are weakly separable by a few morphological traits: flower heads 3 centimeters (1.25 inches) or less in var. *hydrophilum*, and 3 centimeters or more in var. *vaseyi*, and continuous variation in achene size with slightly larger achenes in var. *hydrophilum*. Jepson (1925) and Howell (1949) did not distinguish the Suisun and Mt. Tamalpais *populations* as distinct varieties, treating both as a single variety of *Cirsium vaseyi*. Munz (1959) separated taxa equivalent to *C. hydrophilum* var. *hydrophilum* from var. *vaseyi* and *Cirsium douglasii* by the presence of a fringe of tiny spines along the margins of upper stem leaves and bracts in var. *hydrophilum*. Otherwise, the two varieties of *C. hydrophilum* are distinguished mostly by

ecology (coastal mountain serpentine seep versus *brackish tidal marsh*) and geography (Mt. Tamalpais versus Suisun Marsh).

Cirsium hydrophilum is closely related to two other wetland thistles, the widespread *Cirsium douglasii* DC. (swamp thistle), which also occurs around San Francisco Bay, and *Cirsium mohavense* (Mohave thistle), which is restricted to wet habitats within portions of the Great Basin floristic province (Mohave Desert, east of the Sierra Nevada; Keil and Turner 1993). *Cirsium hydrophilum* can be distinguished from *Cirsium douglasii* mainly by the persistent covering of white, felt-like hairs on both the upper and lower sides of the leaves of *Cirsium douglasii*.

Cirsium hydrophilum resembles several other thistles that occur in wetlands, but only one is likely to occur near or in the same *brackish tidal marsh* habitat in Suisun Marsh. *Cirsium vulgare* (bull thistle), a European weed, is generally found in physically disturbed *marsh* locations where soil *salinity* is low. Plants identified as *C. vulgare*, but with traits intermediate between *C. vulgare* and *C. hydrophilum*, have been reported (Horenstein *in litt.* 1987), and the possibility of hybrid intermediates has been noted (U.S. Fish and Wildlife Service 1997a); no verified specimens of these hybrids have been collected. In mixed local large *populations* of *Cirsium vulgare* and *Cirsium hydrophilum* at Rush Ranch (Suisun Marsh), no intermediate thistles were found (B. Grewell and P. Baye pers. observ. 2000). However, hybridization is not uncommon in thistles (Wells 1983, Dabydeen 1987, Keil and Turner 1993). *Cirsium vulgare* can be distinguished from *C. hydrophilum* within the limited range of *C. hydrophilum* in Suisun Marsh by several useful field characteristics summarized in **Table II-1**.

Table II-1. Summary of field characters for discrimination between *Cirsium vulgare* and *Cirsium hydrophilum* var. *hydrophilum* populations found in Suisun Marsh, Solano County, California (adapted from Keil and Turner 1993, with additions by B. Grewell and P. Baye.)

Trait	<i>Cirsium vulgare</i>	<i>Cirsium hydrophilum</i>
upper leaf surface, basal leaves	coarsely hairy to bristly and dull in maturity	glabrate (few hairs) to glabrous (hairless) in maturity, lacking bristles, somewhat glossy to glossy
lower leaf surface, basal leaves	thin covering of short woolly hairs, appearing pale green	thick covering of long white cob-webby to woolly hairs, appearing white
rosettes	low number of leaves, most large and few-lobed	large number of leaves, continuous size range, mostly with many lobes
stems	with well-developed wings extending from leaf bases; wings strongly spiny	weakly developed or lacking spiny wings
leaf lobes	straight, parallel edges; spines thicker, longer, harder than <i>C. hydrophilum</i>	curved edges; spines more slender, shorter, less hard than <i>C. vulgare</i>
flowerheads	wide at top of egg-shaped head	tightly constricted at narrow top of egg-shaped head
“seeds” (dry fruits)	tan to brown, with thin walls, dull surface, frequently attached to <i>pappus</i> after dispersal	black to dark brown, thick walls, glossy surface, soon detached from <i>pappus</i> before, during, or after dispersal

Other wetland thistles of the San Francisco Bay area that somewhat resemble *Cirsium hydrophilum* include *Cirsium fontinale*, *Cirsium brevistylum*, and *Cirsium andrewsii*. *Cirsium fontinale* has nodding flower heads in contrast with the erect flowerheads of *Cirsium hydrophilum*. *Cirsium brevistylum* and *Cirsium andrewsii* have flowerheads held above clusters of leafy bracts, while *Cirsium hydrophilum* has flowerheads held immediately above a single leaf, but not clusters of leafy bracts (Keil and Turner 1993).

3) Population Trends and Distribution

Historical distribution. There is scarce information on the historical distribution of *Cirsium hydrophilum*. There are no locality descriptions in older regional floras (Jepson 1911, Greene 1894) or herbarium records more specific than “Suisun Marsh(es),” which suggests that it probably did not occur outside the Suisun Marsh area. No records of any form of *C. hydrophilum* occur between the Mount Tamalpais serpentine seep *population* of var. *vaseyi* and the *tidal marsh populations* of var. *hydrophilum* in Suisun Marsh, despite abundant *brackish tidal marsh* habitat along the Petaluma River, Sonoma Creek, and Napa River. One description of the species’ distribution by Greene (for the synonym *Carduus hydrophilus* Greene; Greene 1894) indicates that it was formerly a common plant within the Suisun Marsh region in the late 19th century before *marsh* reclamation prevailed: “Very common in the *brackish* marshes of Suisun Bay, California, where it grows within reach of *tide* water, and is associated with the equally local *Cicuta bolanderi* [synonym: *Cicuta maculata* var. *bolanderi*]....” Subsequent range descriptions (Jepson 1911, 1925; Mason 1957; Munz 1959) do not indicate frequency or range within the *marsh*. It is likely that the elimination of habitat caused by extensive *dike* construction between the 1870s and 1930s in Suisun Marsh (Thompson and Dutra 1983) caused a major decline in species abundance and distribution.

Current distribution. Since the time of listing and in the absence of recent surveys, the species is thought to be present at the two sites known prior to listing (Peytonia Slough Ecological Reserve and Rush Ranch), plus upper Hill Slough and the Joice Island portion of Grizzly Island Wildlife Area, both in Suisun Marsh (California Natural Diversity Database 2006) (**Figure II-2**); however the colonies at Rush Ranch and the colonies at Joice Island, which are at the eastern end of Rush Ranch have generally been interpreted as one *population* (B. Grewell pers. comm. 2000), for a total of three *populations*. Potential habitat exists on private land directly adjacent to the three known *populations* on California Department of Fish and Game and Solano Land Trust properties. The status of the species on private land is unknown.

Peytonia Slough, is a small *population* that fluctuates near extirpation. That *population* was in significant decline in the 1990s, reduced to a single plant in 1996, and totaling only 18 to 30 plants between 1992 and 1994 (B. Grewell pers. comm. 2000). Additionally, it is not known whether a 2001 fire may have affected or eliminated this *population* (Grewell pers. comm. 2007). The other localities are narrowly associated with large pre-historical *tidal brackish marsh* remnants in northwestern Suisun Marsh: Rush Ranch (in the vicinity of Cutoff Slough and First Mallard Branch) and Rush Ranch/Joice Island (in the vicinity of Second Mallard Branch). The Rush Ranch and Rush Ranch-Joice Island *population* consists of numerous discrete colonies totaling hundreds of plants to a few thousand, but these were also in decline during most of the

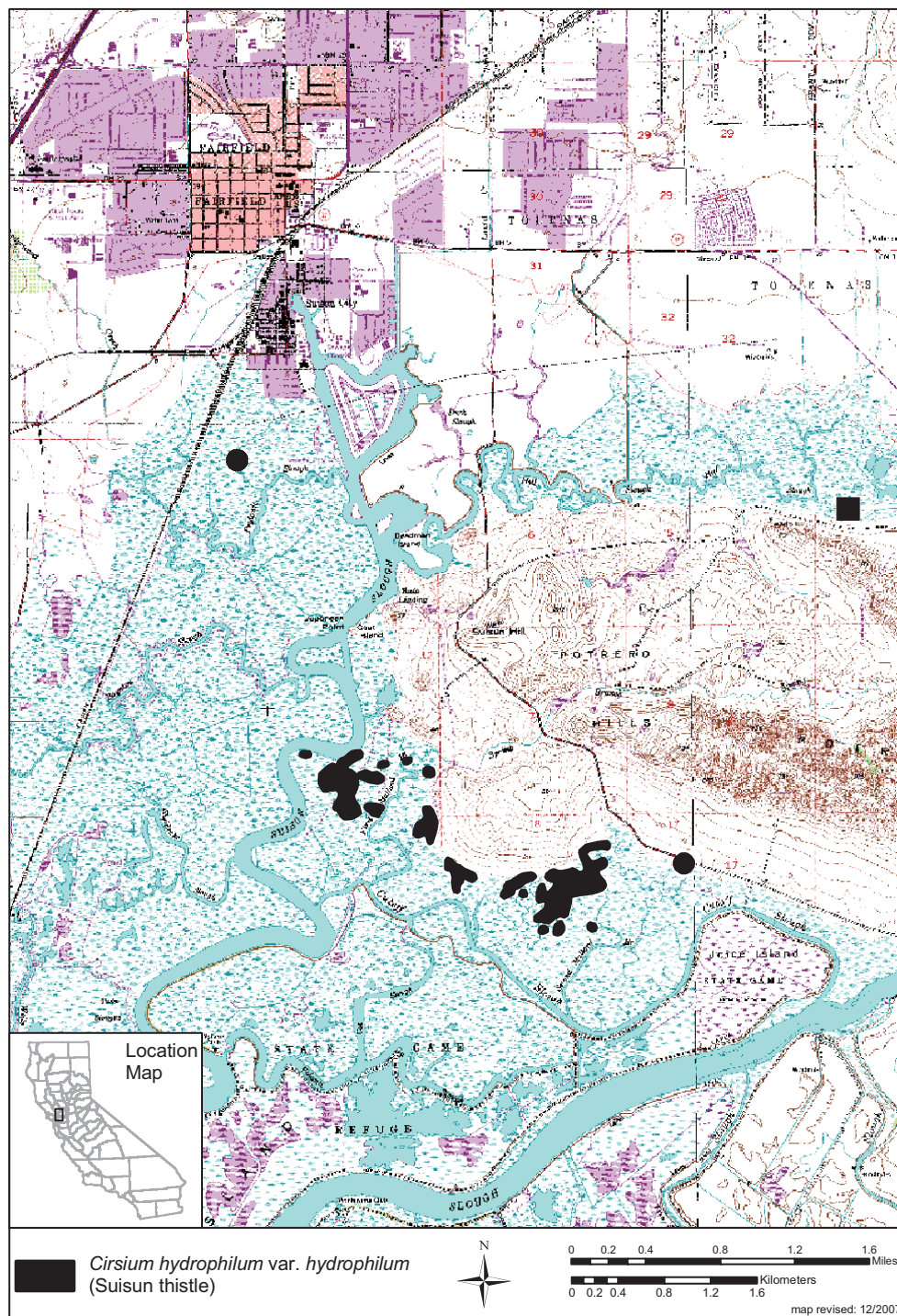


Figure II-2.

Figure II-2. Distribution of *Cirsium hydrophilum* var. *hydrophilum*

1990s (B. Grewell pers. comm. 2000). The *population* in a fully *tidal* area in the upper reaches of Hill Slough was discovered in June 2007 and was estimated at 10 plants (Estrella *in litt.* 2009).

4) *Life History and Ecology*

Reproduction. *Cirsium hydrophilum* is monocarpic, dying after one year of seed reproduction. Its vegetative period is usually one year (biennial), but if small vegetative plant size or unfavorable environmental conditions delay flowering, it may regenerate from the central root crown for more than one year. Flowering occurs throughout the summer in most years, and continues through production of ripe seedheads.

Pollination ecology of *Cirsium hydrophilum* has not been studied, but field observations indicate that thistle colonies in the *marsh* (both native and *non-native* species) attract large swarms of bees (species undetermined) that visit and apparently pollinate the flowers. Bees working thistle colonies are otherwise infrequent in the *tidal marsh* (B. Grewell and P. Baye pers. observ. 2000), although bees commonly act as pollinators of other thistles (Keddy and Keddy 1984, Proctor *et al.* 1996). The abundance of bees pollinating thistles in the *tidal marsh* is probably related to the abundance of potential nest sites and primary nectar/pollen foraging sources in adjacent *uplands*. At Rush Ranch (Suisun Marsh), bees are common and active in extensive stands of *invasive non-native* star-thistles (*Centaurea solstitialis*, *C. calcitrapa*) short distances from *Cirsium hydrophilum* var. *hydrophilum* colonies. The dispersion pattern of *Cirsium hydrophilum* (California Department of Water Resources *in litt.* 1996) in discrete colonies or clusters of small patches suggests there may be limited seed dispersal.

The reproductive output of individual plants and colonies of *Cirsium hydrophilum* var. *hydrophilum* has not been quantified. No quantitative data are available on seed set, seed abortion, or seed predation. Individual branched plants may produce hundreds of seedheads. Seedheads observed in July 2000 ranged from 3 to 15 ripe seeds per seedhead, but many contained all aborted seeds, and some were found with larvae engaged in active seed predation (P. Baye unpubl. data 2000). Soil core samples indicate that soil *seed bank* density of the closely related *C. hydrophilum* var. *vaseyi* may be significant, but the longevity of buried dormant seed in wetland soils is unknown. Cypsela walls (equivalent to seed coats) are hard, and artificially stored seed of *Cirsium hydrophilum* var. *vaseyi* has retained high viability for at least five years (J. Herr pers. comm. 1998). Other thistle species with similar life histories also have persistent soil *seed banks* (Clark and Wilson 1994, Cavers *et al.* 1998). These comparative data with other *Cirsium* species, particularly *Cirsium hydrophilum* var. *vaseyi*, suggest the likelihood of a persistent soil *seed bank* for *Cirsium hydrophilum* var. *hydrophilum*.

Predation. Plant-eating insects can significantly limit seed production and plant *demography* as seen in several other *Cirsium* species (Louda and Potvin 1995, Palmisano and Fox 1997, Rose *et al.* 2005). The introduced thistle weevil (*Rhinocyllus conicus*) has been documented in the Rush Ranch *population* of *Cirsium hydrophilum* where many flowers contained weevil larvae and no seeds (California Department of Water Resources *in litt.* 1996). Louda *et al.* (2003) found that two introduced weevil species (*Rhinocyllus conicus* and *Larinus planus* [Canada thistle bud weevil]) caused *population* decline in native thistle species in the central prairie states. The same

year, L.C. Lee and Associates found *R. conicus* present on *Cirsium hydrophilum* var. *hydrophilum* at Rush Ranch. This weevil destroyed about 15 percent of *viable* seeds produced by the closely related *Cirsium hydrophilum* var. *vaseyi* in serpentine seep habitats, but only early in the flowering season before the end of June. Late flowers escaped predation by the weevil (J. Herr pers. comm. 1998). If this seasonal window applies to Suisun Marsh *populations*, the impact on reproductive output of *Cirsium hydrophilum* may not be highly significant. Flowering time in Suisun Marsh varies with climate, ranging from June to July. This implies the potential for significant weevil impacts at least in some years (B. Grewell pers. comm. 2000).

In addition, larvae of the Mylitta crescent butterfly (*Phyciodes mylitta*) were found to damage vegetative plants of *Cirsium hydrophilum* (California Department of Water Resources *in litt.* 1996). Seeds of *Cirsium hydrophilum* may be subject to pre- and post-dispersal predation, as in other thistle species (Harper 1977), but no data are available. Though documented in the listing rule to have occurred previously at Rush Ranch, *Phyciodes mylitta* caterpillars were not located there during the L.C. Lee and Associates study (LCLA 2003). The rare *endemic* Suisun song sparrow (*Melospiza melodia maxillaris*) is a potential predator of thistle seed, as are common goldfinches. Rodents are also likely seed predators (Klinkhamer and de Jong 1993, Palmisano and Fox 1997); mice that inhabit or visit *tidal* marshes (*e.g.*, salt *marsh* harvest mice, western harvest mice, house mice) may reduce *seed bank* size. The significance of post-dispersal seed predation on reproductive success is unknown.

No information is available on fungal diseases affecting reproduction of *Cirsium hydrophilum*. No parasitism of *Cirsium hydrophilum* by the salt *marsh* plant *Cuscuta salina* (salt *marsh* dodder) has been reported.

Plumed cypsalae (seed-like dry fruits similar to an achene) of thistles are adapted to wind dispersal. The relatively thick-walled, heavy “seeds” of *Cirsium hydrophilum* var. *hydrophilum*, however, readily detach from the plumed *pappus*, sometimes before it disperses (P. Baye pers. observ. 2000). Dispersal patterns of *Cirsium hydrophilum* var. *hydrophilum* seed, therefore, may not necessarily be comparable to those of other thistles with light seeds and persistent *pappus* attachment. There is no evidence of successful long-distance dispersal and colonization of *Cirsium hydrophilum* var. *hydrophilum*. A majority of seed disperses short distances from parent plants. All new colonies detected since listing have been clustered around known *populations* in Suisun Marsh (B. Grewell pers. comm. 2000). The height of the point of seed release has a large effect on dispersal distances of plumed seeds (Harper 1977). The relatively tall stature of *C. hydrophilum* compared with most other associated broadleaf *tidal marsh* plants, combined with the flat topography of the *marsh* and plumed seeds, suggests the potential for long-distance dispersal of those seeds with persistent attached *pappus*. The smooth seed coat of *Cirsium hydrophilum* makes dispersal by attachment to animal fur or feathers unlikely.

Specific conditions for germination and establishment of *Cirsium hydrophilum* var. *hydrophilum* are not known, but field observations suggest they are associated with small gaps or sparsely vegetated areas within the *marsh* plain. Most seedlings of this species established in *Distichlis spicata* (saltgrass)-dominated *brackish tidal marsh* vegetation in the early 1990s, years of relatively high local abundance (B. Grewell pers. comm. 2000). Like most *tidal marsh* species, germination presumably depends on periods of very low *marsh salinity* in winter and early

spring (Woodell 1985). Thus, conditions that promote favorable germination may differ from those that maintain favorable seedling habitat structure (*i.e.*, small gaps or locally sparse vegetation cover established by temporary harsh or disturbed conditions, or species interactions).

5) *Habitat Characteristics/Ecosystem*

Habitat and environmental conditions. *Cirsium hydrophilum* is associated with the upper intertidal *marsh* plain near the smallest branches of natural small *tidal* creeks, banks, ditches, and *marsh* edges that are very infrequently flooded (B. Grewell pers. comm. 2000). Artificial ditch edges and natural creek bank habitats are similar in size, form, and vegetation, but ditches are less stable and more prone to invasion by *non-native* plants. Creek bank edges are typically slightly better drained than other portions of the *marsh* plain. All *Cirsium hydrophilum* populations today occur in peaty organic *marsh* soils, old bay muds of fine estuarine *sediments* (silty clays) with relatively high organic content in the upper horizons, and increasing mineral content with depth (Joice series soils). The soil requirements of the species have not been determined, but they are not known to occur in recently deposited bay muds with lower organic content. It is not known whether the *taxon's* reduced range is due to limitations associated with dispersal, colonization potential, competition, or to specific soil requirements of the species.

Little is known about the *salinity* tolerance of *Cirsium hydrophilum* var. *hydrophilum*. However, it is known to be restricted to *freshwater*-influenced *brackish* marshes, and is absent in the *freshwater tidal* marshes of the west delta and the *tidalsalt* marshes of central San Pablo Bay to the west. Experimental determination of growth and reproduction responses to soil *salinity* is needed to predict the physiological and ecological limits of *Cirsium hydrophilum* var. *hydrophilum*. More complex ecological responses of this *brackish tidal marsh* plant to *salinity* should be determined by its growth in mixed vegetation composed of associated species in variable *salinity* conditions, including sequences of fresh and saline pulses (Howard and Mendelssohn 1999). The absence of *Cirsium hydrophilum* var. *hydrophilum* in west delta marshes may indicate an inability to compete successfully in the tall, dense vegetation of *tidal freshwater marsh* and woody *riparian* thickets. The decline in abundance during a period of above-average rainfall and below average *marsh salinity* is consistent with this hypothesis (B. Grewell pers. comm. 2000). During this period, *freshwater marsh* species increased in relative abundance in Suisun Marsh (P. Baye pers. observ. 1996-1998).

Seedling habitat of *Cirsium hydrophilum* has not been studied. Juvenile plants are found in relatively dense cover of *Distichlis spicata* (saltgrass) and even *Juncus balticus* (wire rush), but seedlings may require gaps in vegetation or sparse areas to establish. Seedlings and juveniles have been found in the vegetation gaps left by large dead plants after exhaustive seed production (P. Baye unpubl. data 2000). Years of high rainfall and concomitant dense growth of tall *brackish marsh* vegetation have been observed to correspond with declines in seedling establishment (B. Grewell pers. comm. 2000). Dense patches of *invasive* *Lepidium latifolium* (perennial pepperweed) appear to displace *tidal marsh* vegetation positively associated with *Cirsium hydrophilum* (B. Grewell pers. comm. 2000; P. Baye pers. observ. 1994-1998). Potential seedling habitat in *brackish marsh* may be provided by vegetation dieback associated with growth of the parasitic *Cuscuta salina* var. *major* (B. Grewell pers. comm. 2000), or

episodes of high soil *salinity* in the *tidal marsh* plain. Thus, temporary harsh, adverse growing conditions for mature plants may be important in regenerating seedling habitat.

Plant associations. *Cirsium hydrophilum* var. *hydrophilum* is associated with various *tidal brackish marsh* plant species of the middle and high *marsh* zones. The earliest information on plant associations was provided by Greene (1894), who emphasized its association with the now-rare *Cicuta maculata* var. *bolanderi* (Bolander's spotted water-hemlock), today seldom if ever found in close association with *Cirsium hydrophilum* var. *hydrophilum*. Little else is known about changes in historical vegetation associated with *Cirsium hydrophilum* var. *hydrophilum*.

The local species composition and relative abundance of *marsh* vegetation in Suisun Marsh is highly variable, and is apparently influenced by soil *salinity* and drainage. *Tidal marsh* plant associations of *Cirsium hydrophilum* var. *hydrophilum* appear to vary with climate cycles. During the relatively drier years of the early 1990s, when *Cirsium hydrophilum* var. *hydrophilum* grew in local relative abundance, low *Distichlis spicata*-dominated vegetation was most often associated with both mature stands and seedling colonies. During the wetter years of the late 1990s, during the period of decline of *Cirsium hydrophilum* var. *hydrophilum* abundance, *Distichlis* associations were largely displaced with dense, tall stands dominated by *Juncus balticus* (Baltic rush), *Scirpus americanus* (Olney's bulrush), and *Lepidium latifolium* (Grewell *et al.* 1999; B. Grewell pers. comm. 2000). Native *tidal marsh* plant species frequently associated with *Cirsium hydrophilum* include *Distichlis spicata*, *Sarcocornia pacifica* (pickleweed), salt rushes of the *Juncus balticus*/*J. lesueurii* complex, *Frankenia salina* (alkali-heath), *Scirpus americanus* (threesquare bulrush), *Potentilla anserina* (silverweed), and *Jaumea carnosa* (fleshy jaumea). The frequency of association does not necessarily imply long-term coexistence, however. Expansion of rush colonies appears to smother seedlings and juvenile thistles in the *tidal marsh* (P. Baye pers. observ. 2000). The rare and endangered *Cordylanthus mollis* ssp. *mollis* (soft bird's-beak) has been associated with *Cirsium hydrophilum* in at least one locality. At Peytonia Slough, plant species growing in the vicinity of *Cirsium hydrophilum* included *Euthamia* (= *Solidago*) *occidentalis* (western goldenrod), *Calystegia sepium* (presumably ssp. *limnophila*; morning-glory), *Oenanthe sarmentosa* (water celery), *Triglochin* spp. (sea-arrowgrass), and *Grindelia camporum* (giant gumplant), *Grindelia stricta* var. *angustifolia* (Suisun gumplant) (Horenstein *in litt.* 1987, California Natural Diversity Database 1997). *Non-native* plant species commonly associated with *Cirsium hydrophilum* include *Lepidium latifolium*, *Cirsium vulgare*, *Sonchus* spp. (e.g., *Sonchus asper* [spiny sowthistle], *Sonchus oleraceus* [common sowthistle]), and *Apium graveolens* (wild celery).

6) Critical Habitat

A final rule designating critical habitat for this species was published April 12, 2007 (U.S. Fish and Wildlife Service 2007a).

7) Reasons for Decline and Threats to Survival

Most species covered in this draft recovery plan are threatened by similar factors because they occupy the same *tidal marsh* ecosystem. These general threats, faced by all covered species, are

discussed in greater detail in the Introduction section of this draft recovery plan (section I.D.). Specific threats to *Cirsium hydrophilum* var. *hydrophilum* are described below.

The fundamental cause of the decline of *Cirsium hydrophilum* var. *hydrophilum* from a locally common to very rare plant was the historical diking of almost all of Suisun Marsh and the conversion of extensive *tidal brackish marsh* to non-*tidal* wetlands (Atwater *et al.* 1979, Dedrick 1989). *Tidalmarsh* area in Suisun Marsh was reduced from 29,000 hectares (71,100 acres; ca. 1850) to less than 4,000 hectares (9,500 acres; Dedrick 1989). Most consists of narrow strips of *marsh* prograded between the edges of *dikes* and the banks of narrowed *tidal sloughs*; relatively little of the pre-historic *marsh* remains. These strip marshes usually support minimal or no *tidal* creek or high *marsh/upland* transition zone habitat suitable for *Cirsium hydrophilum* var. *hydrophilum*. The radical reduction in habitat area and *population* size left *Cirsium hydrophilum* var. *hydrophilum* much more vulnerable to formerly minor threats, such as seed predation/herbivory by insects and invasion by *non-native* vegetation.

Rapid invasion of *brackish tidal marsh* by *Lepidium latifolium* is a very significant threat to the persistence of *Cirsium hydrophilum* var. *hydrophilum* colonies. *Lepidium latifolium* can readily invade both *diked* and *tidal brackish* marshes with low *salinity* during the growing season, and its colonies are especially dense and vigorous in better-drained *marsh* areas where *Cirsium hydrophilum* var. *hydrophilum* is most likely to occur. *Lepidium latifolium* is especially *invasive* on physically disturbed *marsh* soils and where vegetation cover has been reduced. It can permanently establish a continuous leaf canopy, eliminating the vegetation gaps that may be essential for seedling establishment of *Cirsium hydrophilum* var. *hydrophilum*. Dense, tall stands of *Lepidium latifolium* appear to inhibit survival and growth of juvenile thistles as well. Colonies of *Cirsium hydrophilum* var. *hydrophilum* have not been observed to persist in colonies of this *invasive brackish marsh* species (B. Grewell and P. Baye pers. observ.).

Cattle grazing and trampling impacts in *tidal* Suisun marshes are currently remote from most existing colonies of *Cirsium hydrophilum* var. *hydrophilum*, but are locally intensive in unoccupied suitable habitat. This includes areas that support the endangered *Cordylanthus mollis* ssp. *mollis* (soft bird's beak), such as eastern Hill Slough and Peytonia Slough Marshes (B. Grewell pers. comm. 2000). Trampling impacts may be a limiting factor precluding natural colonization or artificial *reintroduction* of *Cirsium hydrophilum* into suitable unoccupied habitat. Evidence of feral hog (*Sus scrofa*) foraging and disturbance has been observed in *Distichlis*-dominated *brackish marsh* within meters of existing *Cirsium hydrophilum* var. *hydrophilum* *populations* at Rush Ranch. Feral hogs pose a significant threat to this critically important *population* (B. Grewell and P. Baye pers. observ. 2000). Limited feral hog hunting has been allowed in portions of Suisun Marsh but a regional-scale eradication effort should be coordinated with California Department of Fish and Game to decrease impact on habitat for sensitive plants.

The historical *population* of *Cirsium hydrophilum* var. *hydrophilum* at Peytonia Slough was reportedly impacted by Suisun Slough dredging activities that altered *tidal* creek *hydrology* for urban drainage and flood control in Suisun City (B. Grewell pers. comm. 2000). Increased residential and commercial development in the expanding Fairfield/Suisun City areas could result in increased urban runoff, *freshwater* discharges from stormwater and wastewater outflows, and adverse hydrological impacts resulting from additional flood control public works

projects. Sustained high levels of nonsaline urban wastewater discharges into Suisun Slough could cause intensive conversion from relatively saline to *freshwater brackish marsh* vegetation. Such conversion could eliminate suitable habitat in the last remaining major *population* near Rush Ranch, resulting in near extinction of the species.

The California Department of Water Resources operates *salinity* control gates at Montezuma Slough to meet artificially low and stable channel water *salinity* standards established to protect water quality for waterfowl *marsh* management (State Water Resources Control Board 1999). Operation of the gates lowers the *salinity* of the *marsh* upstream, and incidentally raises *tidal* elevations on the order of centimeters (Suisun Ecological Workgroup 2001). Preliminary evidence suggests that the altered *salinity* and *tidal* regime may subtly, but significantly, threaten long-term survival of *Cirsium hydrophilum* var. *hydrophilum*. Variation in *salinity* and waterlogging of *marsh* soils over climate cycles causes periodic shifts in structure and composition of the Suisun Marsh *brackish marsh* vegetation resulting in growth inhibition or dieback of more salt-sensitive species. Expansion of low-growing salt-tolerant plants prevails during drought periods, and the reverse occurs in series of wet years. The potential for vegetation gaps to develop apparently increases during environmentally harsh periods of low rainfall and relatively high *salinity* in the *tidal marsh*. Species interactions in California *tidal marsh* plant communities, both positive and negative, are probably mediated by this fluctuating environment (Callaway *et al.* 1990, Callaway 1994, Callaway and Sabraw 1994, Kuhn and Zedler 1997). Water management to enforce artificially low channel *salinity* during droughts, particularly during the summer, is likely to provide a *competitive* advantage to more robust salt-sensitive, *freshwater*-preference *marsh* vegetation, and reduce sub-habitats needed by fugitive gap-colonizing species. *Cirsium hydrophilum* var. *hydrophilum* colonies have been observed to decline during above-average rainfall years (California Department of Water Resources *in litt.* 1996, B. Grewell pers. comm. 2000). Long-term data are needed to clarify this phenomenon and track long-term responses of *Cirsium hydrophilum* *populations* to fluctuations in *marsh salinity* and *tidal* regimes.

Conversely, persistently elevated salinities caused by diversion of *freshwater* outflows from the west delta (Sacramento-San Joaquin Rivers) could also cause conversion to more saline *tidal brackish marsh* and inhibit seedling establishment of *Cirsium hydrophilum* var. *hydrophilum*, causing long-term *population* decline. Of the two potential trends, artificial *salinity* by water management is probably the greater current threat, since Bay-Delta water quality standards have adopted a delta outflow-based approach that maintains a seasonal 2 parts per thousand *salinity* zone (X2 isohaline) within Suisun Bay (State Water Resources Control Board 1999). Even this X2 standard probably reduces Suisun Marsh *salinity* fluctuations below those that prevailed prior to *marsh* reclamation and water diversions.

With strongly reduced modern *populations* of *Cirsium hydrophilum*, and relatively larger surrounding *populations* of *non-native* *Cirsium vulgare*, there is a risk that either competitive displacement, interspecific hybridization and assimilation, or both, could corrupt the *genetic* integrity or *population* viability of *Cirsium hydrophilum* var. *hydrophilum*. Warwick *et al.* (1989) have shown that *populations* of the thistle *Carduus nutans* have been assimilated in *hybrid swarms* involving *Carduus acanthoides*. Some preliminary morphological evidence of hybridization between *Cirsium hydrophilum* and *Cirsium vulgare* has been reported (J.

Horenstein *in litt.* 1987, California Natural Diversity Database 1997), but has not been confirmed by more recent field observations (B. Grewell and P. Baye 2000). Scientific reviews have confirmed the threat to rare plants posed by *genetic* assimilation in *hybrid swarms* (Rieseberg 1991, Levin *et al.* 1996). Even in the absence of hybridization, “pollen swamping” can lower the fitness of insular *populations* of rare species by reducing successful fertilization and seed set (Levin *et al.* 1996).

All three *populations* of *Cirsium hydrophilum* var. *hydrophilum* occur on conservation lands owned and managed by the California Department of Fish and Game or the Solano Land Trust. However, part of the Peytonia Slough *population* may occur on privately owned lands. Public lands are protected from urbanization and agricultural conversion, but many activities that are either unregulated or weakly regulated (*e.g.*, mowing, grazing, ditching) may degrade wetland habitat on privately owned lands. Wetlands owned by the California Department of Fish and Game have been managed for waterfowl hunting in the Suisun Marsh, and some remnant *tidal* marshes were considered for conversion to non-*tidal* waterfowl managed marshes as recently as the early 1990s. Wetland management practices in Suisun Marsh were in partial non-compliance with Endangered Species Act requirements in the 1990s (U.S. Fish and Wildlife Service, file information), which illustrates the possibility of ongoing threats even on protected lands. Conversion of *tidal marsh* to non-*tidal marsh* is currently unlikely to be permitted, but remains a threat because of variability in compliance and enforcement of endangered species regulations.

As described under Life History and Ecology above, pre- and post- dispersal seed predation and *rosette* herbivory are a threat to this species’ survival. The introduced thistle weevil (*Rhinocyllus conicus*) has been documented in the Rush Ranch *population* of *Cirsium hydrophilum* var. *hydrophilum* and may be negatively affecting seed production. Louda *et al.* (2003) found that two introduced weevil species (*Rhinocyllus conicus* and *Larinus planus* [Canada thistle bud weevil]) caused *population* decline in native thistle species in the central prairie states. LCLA (2003) found *Rhinocyllus conicus* present on *Cirsium hydrophilum* var. *hydrophilum* at Rush Ranch. *Phyciodes mylitta* caterpillars, collected on a *population* of *Cirsium hydrophilum* var. *hydrophilum* in September 1996, have caused significant damage to the *rosettes* of plants that will flower the following year (U.S. Fish and Wildlife Service 1997a). Though documented in the listing rule to have occurred previously at Rush Ranch, *Phyciodes mylitta* caterpillars were not located there during LCLA’s 2003 study. They did not collect sufficient data to assess whether *Rhinocyllus conicus* or *Phyciodes mylitta* pose a significant threat to *Cirsium hydrophilum* var. *hydrophilum*. Additional research is necessary to better our understanding of these threats to the species. No management is currently occurring at known locations to ameliorate these threats (Grewell pers. comm. 2007).

A fire started by vandals at Peytonia Slough Ecological Reserve in 2001 may have affected or eliminated this *population* of *Cirsium hydrophilum* var. *hydrophilum* (Grewell pers. comm. 2007). There have not been any surveys for the species at either the burned or unburned portions of the Reserve since this fire.

b. Cordylanthus mollis ssp. mollis
(soft bird's-beak)

1) Brief Overview

Cordylanthus mollis ssp. mollis, soft bird's-beak, was listed as endangered in its entire range on November 20, 1997 (U.S. Fish and Wildlife Service 1997a) with a recovery priority number of 9C, based on its subspecific status, moderate degree of threat, and high recovery potential. The "C" ranking indicates some degree of conflict between the conservation needs of the subspecies and economic development (U.S. Fish and Wildlife Service 1983). It is a California State rare plant (California Department of Fish and Game 2005). *Cordylanthus mollis ssp. mollis* grows in the coastal salt marshes and *brackish* marshes from San Pablo Bay to Suisun Bay in Napa, Solano, and Contra Costa counties. The plant also once occurred in Marin and Sonoma counties, but much of its habitat has been lost or fragmented due to *marsh* alteration and development. There are 11 existing occurrences.

2) Description and Taxonomy

Description. *Cordylanthus mollis ssp. mollis* (soft bird's-beak) is an erect *annual* herb in the Orobanchaceae (broomrape) family (**Figure II-3**). Mature plants range from approximately 10 to 40 centimeters (4 to 16 inches) tall. Plants are typically branched from the middle or above. Stems and leaves are gray-green, often purple-tinged, and covered with very fine hairs bearing *glands* as well as longer soft non-*glandular* hairs. Leaves and stems are sparsely to heavily covered with crystals of salt exuded from leaf *glands*. Leaves are typically 1.0 to 2.5 centimeters (less than 0.5 to 1.5 inches) long, oblong, and may be entire or pinnately lobed (three to seven lobes). The tubular flowers are pale cream to yellowish at the tip, and crowded together in spikes 5.0 to 15.0 centimeters (2 to 6 inches) long. These spikes support about 3 to 30 flowers, each partially covered by a leafy gray-green to purplish lobed bract that resembles a *calyx*. The *calyx* is sheath-like, and encloses most of the *corolla* tube. The *corolla* is densely tomentose (woolly) with yellowish white or greenish yellow lips, and often bears purplish pollinator guides. The upper lip of the *corolla* is beak-like, and encloses the two *stamens* and a *style*; there is also an undeveloped sterile pair of *stamens*. The lower lip of the *corolla* is pouch-like, and divided into three lobes with the middle rolled or folded (Abrams 1951, Chuang and Heckard 1993). The fruit is a capsule, approximately 8 millimeters (0.3 inch) long (Ruygt 1994). Seeds are 2 to 3 mm (0.1 inch) long.

Taxonomy. *Cordylanthus* is a genus of *hemiparasitic annual* herbs closely related to the genus *Orthocarpus* (Chuang and Heckard 1993), a montane genus from which it differs in *calyx* and floral bract traits. *Cordylanthus mollis* A. Gray (1868) was originally based on a type collection by Charles Wright from Mare Island *tidal* marshes (San Pablo Bay, Solano County) in 1855. Synonyms now regarded as invalid include *Adenostegia mollis* Greene (1868) and *Chloropyron molle* Heller (1907). *Cordylanthus mollis* was placed in the subgenus *Hemistegia*, a group of species with *inflorescences* in spikes and an affinity for saline or *alkaline* wetlands. The species *Cordylanthus mollis* was split into two subspecies by Chuang and Heckard (1973), based on geographic variation in spike length, branching pattern, *corolla* hair density, seed size, and hair stiffness. *Cordylanthus mollis ssp. mollis* included all estuarine *populations* and the type of the

species. *Cordylanthus mollis* ssp. *hispidus* (= *Cordylanthus hispidus* Pennell) included the non-tidal inland populations from saline basins and marshes of interior valleys, including saline vernal pools in Solano County near the northeastern reaches of Suisun Marsh. *Cordylanthus mollis* subspecies *mollis* and *hispidus* probably represented coastal and interior forms that have differentiated from an ancestral complex including another interior alkali basin species, *Cordylanthus palmatus* (palmate-bracted bird's-beak, also listed as endangered; Chuang and Heckard 1973).



FIGURE II-3. *Cordylanthus mollis* ssp. *mollis* (Photo Credit Valary Bloom, USFWS)

At the time *Cordylanthus mollis* subsp. *mollis* was listed, the genus *Cordylanthus* was placed in the Scrophulariaceae (figwort family). However, based on molecular systematic studies using DNA sequences of three *plastid genes*, Olmstead *et al.* (2001) transferred the *hemiparasitic* group Castillejiinae, including *Cordylanthus*, to the Orobanchaceae. This systematic treatment will be followed in the upcoming revision of the Jepson Manual.

Additional molecular *phylogenetic* analysis, initiated as part of the above cited studies, indicates that *Cordylanthus* is not a monophyletic genus (Tank and Olmstead 2008, p. 614). In accordance with these findings Tank *et al.* (2009) recognize the genus *Chloropyron* and a previously published name *Chloropyron molle* (A. Gray) A. Heller subsp. *molle* for soft bird's-beak. This combination will also be recognized in the upcoming revision of the Jepson Manual. However, the *taxon* continues to be called

Cordylanthus mollis ssp. *mollis* on the Federal List of Threatened and Endangered Wildlife and Plants (List) pursuant to the Endangered Species Act (Act) (16 U.S.C. 1531 et seq.), and this is the name that will be used in this recovery plan.

Cordylanthus mollis ssp. *hispidus* is distinguished from ssp. *mollis* by its pronounced bristly stem and leaf hairs, and its growth habit of branching strongly from the base of the plant. The flowers of ssp. *hispidus* are sparsely hairy, not densely tomentose (woolly) as in ssp. *mollis*. The Denverton area, Solano County (Suisun Marsh), includes geographic and ecological links between these two subspecies, and is known to support some *populations* of *Cordylanthus mollis* ssp. *hispidus* in non-tidal alkaline seasonal wetlands (Ruygt 1994). These *populations* near the *estuary* edge potentially provide opportunities for intercrosses between the subspecies, the existence and importance of which are unknown.

Within its range, *Cordylanthus mollis* ssp. *mollis* can be distinguished from two other taxa in the Scrophulariaceae that occur in *brackish tidal* marshes: *Cordylanthus maritimus* ssp. *palustris* (northern salt *marsh* bird's-beak) and *Castilleja ambigua* (Johnny-nip, salt marsh owl's-clover). Both are also *hemiparasitic annual* herbs with affinity for saline wetland soil. *Cordylanthus maritimus* ssp. *palustris* has ecological similarities to *Cordylanthus mollis*. It has become very rare in the San Francisco Bay Estuary (restricted to the Petaluma Marsh, Heerdt Marsh, and Richardson Bay, all in Marin County), overlapping only slightly the historical range of *Cordylanthus mollis*. When in flower, *Cordylanthus maritimus* in the San Francisco Bay Estuary is readily distinguished from *Cordylanthus mollis* by its rose-purple and pinkish-white flowers, and the presence of four fully developed *stamens* (not two plus two vestigial *stamens*, as in *Cordylanthus mollis*). The inner bracts of *Cordylanthus maritimus* are notched, not lobed, while the bracts of *Cordylanthus mollis* are pinnately lobed.

Castilleja ambigua (synonym *Orthocarpus castillejoides*) is now very narrowly distributed within the San Francisco Bay Estuary. The only known *population* occurs at Point Pinole *tidal* marshes, with some individuals near *Cordylanthus mollis* colonies. *Castilleja ambigua* occurs in high *tidal marsh* and the upland *ecotone* with relatively low, sparse vegetation cover. The *population* of *Castilleja ambigua* in San Francisco Bay flowers in spring (variably late March to May) before *Cordylanthus mollis*. The bracts and leaves of *Castilleja ambigua* are palmately cleft, not pinnately lobed as in *Cordylanthus mollis*. Although typical *Castilleja ambigua* ssp. *ambigua* has white and yellow flowers like *Cordylanthus mollis*, the Point Pinole *population* of ssp. *ambigua* and other historical San Francisco Bay *populations* have flowers that mature and senesce with a purplish tinge (P. Baye unpubl. data 1997-2000), as do the white-tipped bracts (Chuang and Heckard 1993). In contrast, the bracts of *Cordylanthus mollis* are gray-green or a blend of gray-green and dull dark purplish highlights, and its flowers are creamy yellow or yellowish-green and lack an open beak tip that allows the *stigma* to protrude (Chuang and Heckard 1993).

3) Population Trends and Distribution

Historical distribution. Early California floras describe the range of *Cordylanthus mollis* either from Mare Island/Vallejo alone (Brewer *et al.* 1880, Behr 1888), or Vallejo and Suisun (Greene 1894, Jepson 1911). The western limit of historically verified *populations* extended to the *tidal*

marshes between the Petaluma River and San Rafael (Howell 1949), where it was described as “not common” in 1897. *Cordylanthus mollis* ssp. *mollis* was collected from the Burdell locality (Marin County) along the western Petaluma River as recently as 1966. In its western (Marin County) range, it was locally sympatric (occurring in the same geographical range) with *Cordylanthus maritimus* (Howell 1949). The eastern range of *Cordylanthus mollis* ssp. *mollis* extends to *brackish tidal* marshes at the mouth of the Sacramento River. A *population* west of Antioch Bridge (California Natural Diversity Database 1997) was observed only once and apparently did not persist (B. Grewell pers. comm. 2000). Grewell *et al.* (2003) report a *population* between Pittsburg and Antioch in Contra Costa County.

It is questionable whether the range of *Cordylanthus mollis* ssp. *mollis* actually extended to the salt marshes of San Francisco. Howell *et al.* (1958) include *Cordylanthus mollis* in the flora of San Francisco, based on a single collection by C.C. Parry from 1881 with a label indicating San Francisco as the location. Chuang and Heckard (1973) suggest that this location may refer to the vicinity of San Francisco Bay rather than the county itself. San Francisco marshes were relatively well surveyed floristically compared with other locations, and no other records of *Cordylanthus mollis* in San Francisco were reported by Brandegee (1892), Behr (1888), Greene (1894), Jepson (1901), or other early botanists. Either *Cordylanthus mollis* was *extirpated* very early from San Francisco *tidal* marshes or, more likely, this locality is based on misinterpreted or erroneous specimen labeling.

Current distribution. There are currently 11 *populations* with documented occurrences in nine general areas: Rush Ranch, Hill Slough, Joice Island, Benicia State Recreation Area, Point Pinole, Concord Naval Weapons Station, Fagan Slough, McAvoy Boat Harbor and Denverton (**Figure II-4**). Understanding of the current distribution of *Cordylanthus mollis* ssp. *mollis* is based on limited and opportunistic survey data. No recent comprehensive rangewide status survey has been conducted for *Cordylanthus mollis* ssp. *mollis*. The largest *populations* today are located mostly in old relict *tidal* marshes of Suisun Marsh. The most recent near-comprehensive census was conducted in 2000 (Grewell *et al.* 2003). This census covered Hill Slough Marsh and Rush Ranch, both in Suisun Marsh, Solano County. It also included Benicia State Recreation Area (Solano County) and Fagan Slough Ecological Reserve (Napa County; Grewell *et al.* 2003). The largest *population* was found at Hill Slough Wildlife Area and covered approximately 2 hectares (4.7 acres).

A more recent *population* distribution and status evaluation was conducted in 2004 strictly for reference *populations* at Benicia State Recreation Area and the *populations* at the site of a 2000 experimental *reintroduction* at Rush Ranch (Grewell 2005). The Rush Ranch *population* was estimated to be 95,510 individuals occupying 0.08 hectares (0.2 acres). The estimated *population* at Benicia State Recreation Area had 99,005 individuals, the highest numbers ever recorded for a *population* of *Cordylanthus mollis* ssp. *mollis*, occupying a total area of 0.31 hectares (0.77 acres; Grewell 2005). Many *annual* plants are known to have large fluctuations in *population* sizes among years, and the high numbers recorded in 2004 may be a reflection of this characteristic. It is also possible that this was the most thorough search ever conducted, based on micro-habitats surveyed (Grewell 2005). Although *population* monitoring at Rush Ranch and Benicia State Recreation Area indicated continued *population* growth from 2000 to 2004, seed production of the reintroduced *population* at Rush Ranch plummeted for unknown reasons in

2004 (Grewell 2005). Long-term monitoring of *population* sizes will be more useful in determining viability of the *population* than a single season census.

Each *population* of *Cordylanthus mollis* ssp. *mollis* is composed of many shifting colonies or *subpopulations*. Discrete *populations* consist of widely spaced stands or clusters of colonies with significant dispersal barriers. Most colonies have locally high densities of plants, but some may be diffuse or scattered. *Population* size and distribution are extremely variable among years. *Subpopulations* may fail to appear entirely some years and reappear later. Because of the great variability in *population* size and distribution, short-term (one or two years) estimates of *population* location and size are not meaningful as indices of actual *population* size. *Population* viability, or trends of growth and decline, must be interpreted over a number of years. The area regularly inhabited is also an important measure of the security of the species.

Because colonies may fail to emerge in some years, it can be difficult to determine with confidence when a *population* has become *extirpated*. Sites where the species has not been detected for many years, but where suitable habitat with potentially intact *seed banks* persists, should be interpreted cautiously. Sites where *populations* have only recently declined or which have not been rigorously surveyed may be presumed extant but latent. The size and distribution of *viable seed banks* in *marsh* soils would probably be a more meaningful indicator of *population* size. However, data on soil *seed banks* are not currently available.

In 2000, six of the main *populations* were estimated to contain a total of roughly 300,000 individuals in about 200 patches or stands (Grewell *et al.* 2003). According to some estimates, the important Hill Slough *population* has experienced a persistent decline from 1993 through 1999 and 2001 (Grewell *et al.* 2003, Grewell 2004).

4) Life History and Ecology

Reproduction. *Cordylanthus mollis* ssp. *mollis* is an *annual* plant that evidently regenerates from a persistent dormant *seed bank*. The longevity of the *seed bank* is unknown. However, some colonies have failed to emerge for several years and then reappeared. *Population* densities vary from isolated individuals (fewer than 0.5 per square meter [.05 per square foot] to more than 450 per square meter [42 per square foot]), with densities of 100 to 200 per square meter (10 to 20 per square foot) common (Ruygt 1994).

The netted surface of *Cordylanthus mollis* ssp. *mollis* seeds traps microscopic air pockets, making them buoyant and well-adapted for flotation. Although this trait may enable seeds to disperse long distances on *tidal* currents, dispersion patterns determined by repeated surveys indicate that most dispersal occurs over short distances (Ruygt 1994) on the order of 10 meters [33 feet] or less (Grewell *et al.* 2003). However, studies of dispersal generally are unlikely to detect rare long distance dispersal.

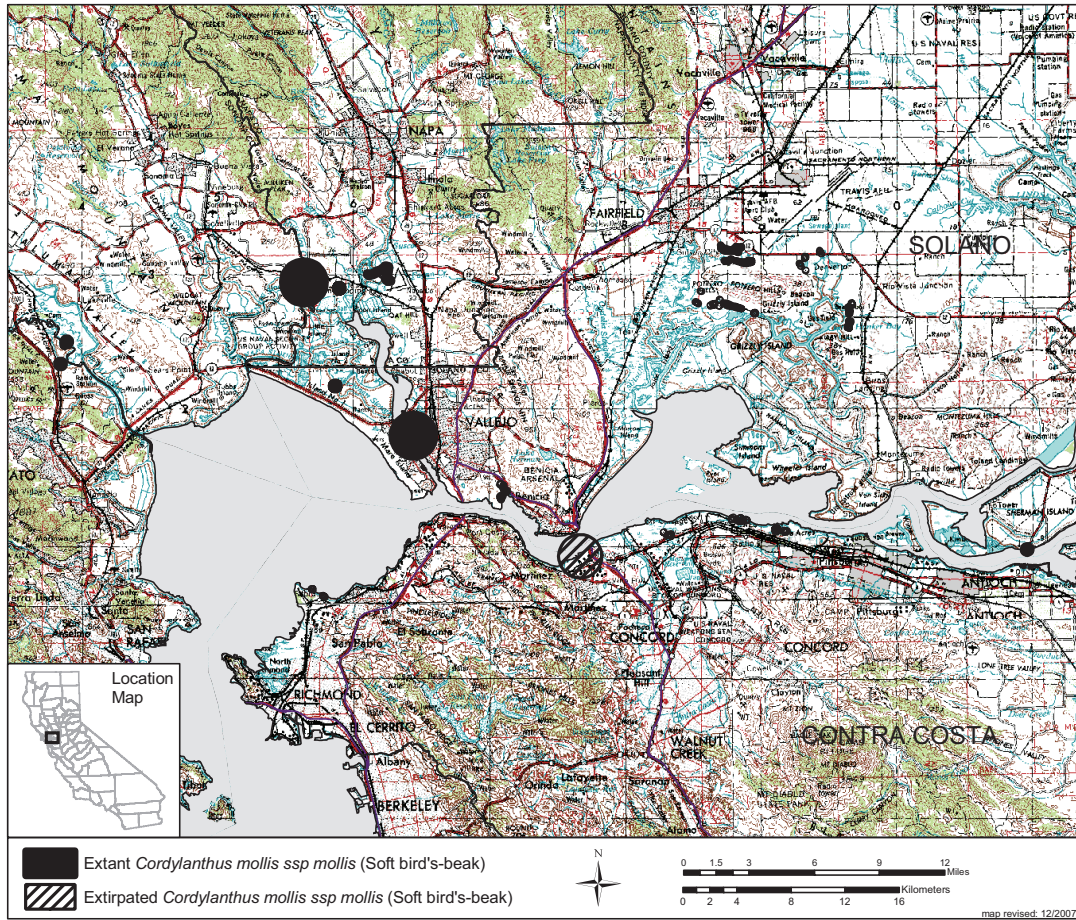


Figure II-4.

Figure II-4. Distribution of *Cordylanthus mollis ssp. mollis*

Seed germination is correlated with fall/winter rainfall, from December to April, and occasionally earlier or later (B. Grewell pers. comm. 2000). Peak germination rates are in February and March (Ruygt 1994). Although most salt *marsh* plants have seed germination tied to periods of *tidal marsh* flooding with low soil *salinity* (Woodell 1985), germination of *Cordylanthus mollis* ssp. *mollis* in Suisun Marsh has been observed to be greatest in areas with extended *tidal hydroperiods* and somewhat higher soil *salinity* (Grewell 2004). However, central areas of *scalds* and other areas with low plant density support fewer *Cordylanthus mollis* ssp. *mollis*, whether due to *salinity* or lack of host plants is unknown.

Hemiparasites, such as *Cordylanthus mollis* ssp. *mollis*, are parasitic plants that have chlorophyll and are capable of some photosynthesis. While *Cordylanthus mollis* ssp. *mollis* plants can survive independently under ideal greenhouse conditions, a host plant (or plants) is needed to survive and reproduce in the wild (Ruygt 1994, Grewell *et al.* 2003). Seedling survival is critically dependent on establishing an early connection with a suitable host plant. The parasitic root connections, called *haustoria*, are short at the seedling stage (less than 5 cm [2 inches]) and rather fragile (Grewell *et al.* 2003). Photosynthate and water are major constituents transferred from the host to the *hemiparasite* via the *haustoria*. *Cordylanthus mollis* ssp. *mollis* sometimes establish *haustorial* connections with other conspecific individuals, and it may be that photosynthate from a host can be transferred indirectly via another intervening plant to one not immediately connected to the host.

Seedlings of *Cordylanthus mollis* ssp. *mollis* will attach to a broad range of hosts, but not all plants make suitable hosts for the species. Known beneficial hosts include many summer-active native species, including *Sarcocornia pacifica* (pickleweed), *Distichlis spicata* (saltgrass), and *Jaumea carnosa* (fleshy jaumea). Winter *annuals* such as *Juncus bufonius* (toad rush) and many *non-native annual* grasses and forbs (e.g., *Polypogon monspeliensis* [annual beard grass], *Hainardia cylindrica* [barbgrass], and *Cotula coronopifolia* [brass-buttons]) do not appear to be suitable hosts because they typically are dying by the time *Cordylanthus mollis* ssp. *mollis* plants need to flower and set seed. Seedlings of *Cordylanthus mollis* ssp. *mollis* suffered increased mortality when they happened to germinate in the near neighborhood of these unsuitable hosts or in an area with low biomass of hosts (Grewell *et al.* 2003). Invasion of *non-native* plants is a threat to *Cordylanthus mollis* ssp. *mollis* both because many *non-natives* are unsuitable hosts and because they may compete with and reduce the density and biomass of native host plants.

Branching and flower development begin as early as May (Ruygt 1994) and continue throughout the summer. Flower production in *Cordylanthus mollis* ssp. *mollis* correlates with the degree of branching and plant size (Ruygt 1994, Grewell *et al.* 2003, Grewell 2004). Fruits and seeds mature from July to November. At Fagan Slough in 1993, flowering reached a peak in late July-early August, and declined strongly by late August. Flowering has been observed to occur, however, as late as November, indicating a significant overlap between flowering and fruiting (seed production) time. Some fruits begin to mature around early July.

Several types of generalist native bees and other potential pollinators have been observed visiting *Cordylanthus mollis* ssp. *mollis* flowers. Bumblebees (*Bombus californicus*, possibly other *Bombus* species [Apidae]) were the most frequent visitors in a study by Ruygt (1994). Leaf cutter bees (*Anthidium edwardsii*: Megachilidae) and a sweat bee (*Halictus tripartitus*:

Halictidae) also were seen visiting flowers, but their significance as pollinators is uncertain. *Anthidium edwardsii* was the most abundant visitor to *Cordylanthus mollis* ssp. *mollis* flowers in a study by Grewell *et al.* (2003), followed by *Lasioglossum* sp., *Halictus* sp. (both Halictidae), and individuals of *Bombus californicus* and *Bombus vosnesenskii*. Other occasional visitors were another native solitary bee (*Melissodes*: Anthophoridae) and a bee fly (Diptera: Bombyliidae). Grewell *et al.* (2003) note the possibility that *non-native* and native flowers of other species may compete for the attention of available pollinators, and specifically referenced yellow star thistle (*Centaurea solstitialis*). The pollinators known to visit *Cordylanthus mollis* ssp. *mollis* are generalists, that is, they will visit a variety of flowers, and could be attracted away by an abundance of another flowering species.

Relatively low numbers of pollinators were observed visiting *Cordylanthus mollis* ssp. *mollis* populations that had high reproductive output at Hill Slough, suggesting some degree of self-pollination. This is consistent with preliminary experimental work in which pollinators were excluded and some seeds were still produced (Ruygt 1994). Nevertheless, the degree to which reproductive output is dependent on or limited by pollinators is uncertain. Grewell *et al.* (2003) believe the species is dependent on insect pollinators for full reproductive output. Parsons and Zedler (1997) found that even a self-compatible population of *Cordylanthus maritimus* ssp. *maritimus* required insect pollinators to achieve high seed set.

Predation. Seed output of at least some *Cordylanthus mollis* ssp. *mollis* populations is strongly constrained by seed predation, or granivory (Ruygt 1994). Insects that feed on flowers, fruits, and seeds caused substantial reduction in fruit and seed set. Salt marsh snout moth larvae (*Lipographus fenestrella*: Pyralidae) caused significant damage to flowers at the large Hill Slough population, and have been inferred to damage populations at Fagan Slough and Joice Island (Ruygt 1994). Seed capsules filled with insect frass are common in Southampton Marsh as well (P. Baye pers. observ. 1997-2000, Grewell *et al.* 2003). Another moth species (initially identified as *Ptycholoma* sp., now thought to be *Saphenista* [Tortricidae]) caused flower damage at the Fagan and Hill Slough populations (Ruygt *in litt.* 1993), and in 2001 was the main seed predator at Hill Slough (Grewell *et al.* 2003). Losses of seed to larval feeding can be very high, with up to 71 percent of flowering branches in a population affected (Ruygt 1994), or mature seed output 50 to 70 percent lower in populations with high moth damage (Grewell *et al.* 2003). Other *Lepidopteran* larvae identified on *Cordylanthus mollis* ssp. *mollis* were the common buckeye butterfly (*Junonia coenia*: Nymphalidae) and another moth, *Perizoma custodiata* (Geometridae; Grewell *et al.* 2003).

Herbivorous insect populations often go through boom and bust cycles, which may not be synchronous among different population locations. This suggests the importance of multiple peripheral populations of *Cordylanthus mollis* ssp. *mollis* that may escape large outbreaks of plant-eating insects. For example, the Rush Ranch reintroduction site was not discovered by seed predators in the first year, and also escaped significant damage in its second year (Grewell *et al.* 2003). The soil seed bank may be important for buffering the long-term effects of seed predation on population viability.

Seed predators themselves have predators, notably vespid wasps (yellowjackets and potter wasps) that search *Cordylanthus mollis* ssp. *mollis* inflorescences for larvae with which to feed

their young (Grewell *et al.* 2003). Preserving and managing nearby native habitat for these and other predators, parasites, and diseases of the seed-damaging species would likely benefit *Cordylanthus mollis* ssp. *mollis* population dynamics.

Seeds may also be subject to predation after they have been shed from the maternal plant. Some granivorous species observed in the vicinity of *Cordylanthus mollis* ssp. *mollis* at the time of seed drop include savannah sparrows (*Passerculus sandwichensis*), western meadowlark (*Sturnella neglecta*), and salt marsh harvest mice (*Reithrodontomys raviventris*; Grewell 2004). It is not suggested that native granivores be controlled. If other threats to *populations* of *Cordylanthus mollis* ssp. *mollis* are reduced or eliminated, the species will likely tolerate the pressure of native granivory without ill affect.

Other species that could affect *Cordylanthus mollis* ssp. *mollis* include rabbits and deer, which are relatively indiscriminate grazers of plants. Unrestricted cattle grazing and trampling also occur in some *populations* (Ruygt 1994), and their effects appear to be harmful to *population* regeneration (California Department of Water Resources *in litt.* 1996, Fiedler *in litt.* 1996). Livestock can spread *non-native* invasive plants. Uprooting of *marsh* soils by feral pigs may also cause at least short-term adverse impacts to adult and *seed bank populations*, but no direct evidence has yet been reported. Soil disturbance, such as digging by pigs, heavy trampling by cattle, and a wide variety of human activities, often facilitates *non-native* weed invasion, which adversely impacts *Cordylanthus mollis* ssp. *mollis* survival.

In spite of its parasitic habit, *Cordylanthus mollis* ssp. *mollis* has beneficial effects on some species in the ecosystem, and has even been called an “ecosystem engineer” (Grewell 2004). Dominant species (*e.g.*, *Sarcocornia*, *Distichlis*) are reduced by *Cordylanthus mollis* ssp. *mollis* presence, and less abundant species are able to increase (*e.g.*, *Atriplex triangularis* [spearscale], *Triglochin maritima* [seaside arrowgrass]), allowing for a more diverse community. *Cordylanthus mollis* ssp. *mollis* also modifies the soil environment where it occurs, causing increased soil oxygenation and lowered soil *salinity* by enhancing translocation of salty water out of the soil. High nutrient content in decomposing *Cordylanthus mollis* ssp. *mollis* plants may further diversify the spatial pattern of soil conditions in the *marsh*. These factors probably result in increased ecosystem diversity when *Cordylanthus mollis* ssp. *mollis* is present (Grewell 2004).

5) Habitat Characteristics/Ecosystem

The principal habitat of *Cordylanthus mollis* ssp. *mollis* is the high *marsh* zone or upper middle *marsh* zone of *brackish* marshes with full *tidal* range (Peinado *et al.* 1994). It is rarely found in non-*tidal* conditions (a single collection is known: L.R. Heckard 4665, JEPS76417). *Cordylanthus mollis* ssp. *mollis* abundance is often greatest in or near the upper *marsh*-upland *ecotone* (Chuang and Heckard 1973, Ruygt 1994). Large, dense patches are sometimes found along the margins of emergent *salt pans*, or *scalds* (Ruygt 1994).

Colonies of *Cordylanthus mollis* ssp. *mollis* may occur on different kinds of soils, including peaty clay-silt *tidal marsh* soils along natural low-relief *levees* of *tidal* creek banks (*e.g.*, Point Pinole, Southampton Marsh; Ruygt 1994), or on primarily mineral alluvial *sediments* at the margins of shallow *salt pans* at the upper *marsh* edge (southwestern Southampton Marsh near

Dillon Point; P. Baye pers. observ. 1997-2000). Plants have been found to colonize *marsh* soils formed on top of artificial fill (Ruygt 1994).

Ruygt (1994) found that soil *salinity* peaked at the margins of barren *scalds* near the upper *marsh* edge. Despite the extreme *salinity* potential of this subhabitat, the edges of these *scalds* may be associated with high local abundance of *Cordylanthus mollis* ssp. *mollis* (Ruygt 1994). The overall geographic range of this species, however, is *freshwater*-influenced, *brackish tidal* marshes of the *estuary*. Only one modern *population* (Point Pinole) is in salt *marsh* vegetation with little *freshwater* influence. Parasitism of neighboring plant roots may buffer soil moisture and *salinity* stresses (Chuang and Heckard 1971).

Studies of the ecologically similar species *Cordylanthus maritimus* indicate that its distribution in salt marshes corresponds with vegetation that is sparse, low, or contains small gaps to enable seedlings to establish in the absence of strong competition and shade. *Cordylanthus maritimus* is negatively correlated with dense, tall, or continuous vegetation patches with low species diversity (Parsons and Zedler 1997, U.S. Fish and Wildlife Service 1985a, Pickart and Miller 1988, Kelly and Fletcher 1994). These habitat traits are broadly applicable to *Cordylanthus mollis* ssp. *mollis* as well, with important exceptions. *Cordylanthus mollis* ssp. *mollis* has been observed in areas of past disturbance where vegetation cover is suppressed, including on old dredge spoils along ditches (Ruygt *in litt.* 1993), old roads, and footpaths (B. Grewell pers. comm. 2000, P. Baye pers. observ. 1997-99.). Vigorous plants in dense patches have also been observed overtopping thick *Sarcocornia pacifica* (pickleweed) vegetation along salt *pan* edges (P. Baye pers. observ. 2000). Environmental and *biotic* factors that cause sparse vegetation patches include *driftlines* (smothering by *tidal* litter deposits; Chapman 1964, Hartman *et al.* 1983, Parsons and Zedler 1997), parasitism by *Cuscuta salina* var. *major* (salt marsh dodder; Grewell *et al.* 2003, Grewell 2004), and low rainfall and *salinity* stress (Allison 1992, Callaway 1994). Variation in soil conditions along upland salt *marsh* edges also appears to influence species distribution, and the density and cover of *tidal marsh* vegetation in the San Francisco Bay Estuary, especially around summer-dry salt *pans* (Baye *et al.* 1999).

Plant associations

Native plant species typically associated with *Cordylanthus mollis* ssp. *mollis* in the *brackish* high *marsh* and upper middle *marsh* zone (*marsh* plain) include dominant species *Sarcocornia pacifica*, *Distichlis spicata*, and *Cuscuta salina* var. *major* (salt marsh dodder), and associates *Frankenia salina* (alkali-heath), *Jaumea carnosa*, *Atriplex triangulari*, *Triglochin maritima*, *Plantago subnuda* (Mexican plantain), *Plantago maritima* (seaside plantain), *Grindelia stricta* var. *angustifolia* (salt marsh gumplant), and *Limonium californicum* (sea-lavender; Ruygt 1994, California Natural Diversity Database 1997, Grewell *et al.* 2003). *Non-native* plants may also be locally abundant associates of *Cordylanthus mollis*, including low *annuals* such as *Hainardia cylindrica*, and tall dominant *perennial* herbs such as *Lepidium latifolium* (perennial peppergrass). *Sarcocornia* and *Distichlis* are host species, and *Cuscuta* was the most closely associated species in the study of Grewell *et al.* (2003).

Although *Cordylanthus mollis* is *hemiparasitic*, the specific plant host-parasite relationships have not been closely studied. Based on studies with other *Cordylanthus* species, the primary

benefit of parasitism appears to be water availability during drought periods (Chuang and Heckard 1971, Vanderweir and Newman 1984). *Cordylanthus* species are generally capable of completing their life-cycles under favorable soil conditions even in the absence of a host, but may require hosts to survive severe soil moisture stress caused by high *salinity* (Chuang and Heckard 1971).

Parasitic *Cuscuta salina* has been observed to parasitize *Cordylanthus mollis* ssp. *mollis* occasionally (Ruygt 1994; P. Baye pers. observ. 1999, 2000; Grewell *et al.* 2003), but it is seldom parasitized as heavily as *Sarcocornia*. *Cuscuta salina* may be of greater indirect benefit to *Cordylanthus mollis* ssp. *mollis* by causing local dieback and vegetation gaps, allowing *annuals* to colonize open patches (Grewell 2004).

6) Critical Habitat

A final rule designating critical habitat for this species was published April 12, 2007 (U.S. Fish and Wildlife Service 2007a).

7) Reasons for Decline and Threats to Survival

Most species covered in this draft recovery plan are threatened by similar factors because they occupy the same *tidal marsh* ecosystem. These general threats, faced by all covered species, are discussed in greater detail in the Introduction section of this draft recovery plan (section I.D.). Specific threats to *Cordylanthus mollis* ssp. *mollis* are described below.

There are many current threats that place *populations* of *Cordylanthus mollis* ssp. *mollis* at risk of local extinction, but the principal cause of the species' current rarity and decline is the extensive loss of its narrow habitat caused by diking of large tracts of *tidal* marshes. Diking for agricultural reclamation destroyed most of the original *tidal* marshes in the northern part of the San Francisco Bay Estuary, reducing *tidal marsh* acreage to approximately 15 percent of historical area overall (Goals Project 1999). Most of this residual *tidal marsh* was formed by recent sedimentation, with very little of the pre-historic *marsh* area actually remaining (Dedrick 1989). Most *populations* of *Cordylanthus mollis* ssp. *mollis* are associated with areas of relict old *tidal* marshes.

The impacts of diking on *Cordylanthus mollis* ssp. *mollis* were probably greater than the total *tidal marsh* loss suggests, because overall loss estimates do not distinguish the subhabitats lost. It is very likely that there was a proportionally larger loss of high and upper middle salt *marsh* zones near the landward edge of *tidal* marshes and along larger *tidal sloughs* (optimal *Cordylanthus* topography and habitat), because *dikes* were normally constructed precisely along these natural shoreline boundaries (Ver Planck 1958, Thompson and Dutra 1983). Large-scale loss of habitat caused by diking and reclamation probably isolated many of the *populations* of *Cordylanthus mollis* ssp. *mollis*, which survived in remnant *tidal marsh* outside of *dikes*. Fragmentation of *populations* increases the likelihood of their local extinction.

The reduction of total habitat area available to *Cordylanthus mollis* ssp. *mollis*, and the isolation of its *populations*, magnifies the impacts of localized threats in remnant habitats. These threats

include *dike* repair and maintenance, ditch maintenance, grading and stabilization activities at *marsh* edges (e.g., Belden's Landing), and cattle grazing (e.g., Hill Slough). Even natural processes such as channel bank erosion (e.g., Point Pinole) can threaten small *populations* that have lost *resilience* because of long-term reduction in their size and extent, and increased dispersal distances to neighboring *populations*. Some impacts, like ditch maintenance, may have both adverse and beneficial effects in different time-scales. Initial disturbance may cause elimination or reduction of small *populations*, but may also open vegetation gaps and create microtopography that favors long-term colonization by *Cordylanthus mollis* ssp. *mollis*. However, vegetation gaps caused by disturbance also invite establishment of *invasive non-native* plants, an adverse effect to *Cordylanthus mollis* ssp. *mollis*. Adverse impacts caused by such disturbances are easily predictable, but it is difficult to predict the likelihood or magnitude of recolonization or *population* increases because of the influence of random factors. Prediction of *population* changes, and detection of impacts on *population* size, is very difficult because of the high natural annual fluctuation of *populations*.

The most significant threats to remaining *Cordylanthus mollis* ssp. *mollis* *populations* are region-wide. One of the most potentially detrimental is the invasion of the middle and upper *brackish tidal marsh* zones by *non-native* *Lepidium latifolium*, a tall clonal herb in the mustard family that establishes in dense stands. *Lepidium latifolium* generally excludes *Cordylanthus mollis* ssp. *mollis*. There are no reports of its *populations* regenerating annually under spreading tall canopies of *Lepidium latifolium*. The invasion of *brackish tidal* marshes by *Lepidium latifolium* has proceeded rapidly in the last two decades. It currently threatens at least portions of *Cordylanthus mollis* ssp. *mollis* *populations* at Rush Ranch and Southampton Marsh where it has spread rapidly in the last decade, particularly in high rainfall years (B. Grewell, P. Baye pers. observ. 1991-1999). Other *invasive* plants threatening the survival of *Cordylanthus mollis* ssp. *mollis* seedlings include the grasses *Hainardia cylindrica* and *Polypogon monspeliensis* (annual beard grass; Grewell 2005). These species serve as inappropriate host plants because they die off before *Cordylanthus mollis* ssp. *mollis* completes its life cycle. The effect of *Cotula coronopifolia* (brass-buttons) may also deserve further examination.

Another potential regional threat to *Cordylanthus mollis* ssp. *mollis* is the large-scale alteration of *salinity* regimes, particularly attempts to stabilize low salinities in the Suisun Marsh to promote water quality standards for selective public beneficial uses. These do not adequately consider the needs of endangered plant species. In recent decades water quality standards for *salinity* in Suisun Marsh emphasized conditions specifically favorable for waterfowl habitat (State Water Resources Control Board 1999). Non-*tidal* flooding of *diked* marshes inevitably results in some evaporative concentration of salts (like salt ponds), and may result in salt accumulation if drainage is poor. Relatively saline Suisun *tidal* water is *brackish* in spring and fall during droughts. It is less saline than San Francisco Bay water, but conducive to producing *hypersaline* conditions after evaporation. Water quality standards were modified in light of broader estuarine ecological considerations (State Water Resources Control Board 1999). Although water quality standards for *salinity* have been modified in western Suisun Marsh to allow for climate-driven fluctuation, the artificially narrow low *salinity* range is still enforced for the eastern Suisun Marsh.

Under natural conditions Suisun Marsh *salinity* would be closely linked with delta outflows and drainages of the Suisun Marsh watershed. In the early 1990s, the California Department of Water Resources constructed and operated tidegates in Montezuma Slough to maintain low summer and fall salinities regardless of delta outflows. Operation of the *salinity* control gates has widespread effects on *tidal marsh* soil and water *salinity*, and even *tidal datums*, in the Suisun Marsh area. Persistent low summer soil *salinity* during high rainfall years favors conversion of middle *tidal marsh* zones to *Scirpus*-dominated vegetation, with concomitant loss of *Sarcocornia-Distichlis* vegetation associated with *Cordylanthus mollis* ssp. *mollis*. During dry years *Sarcocornia-Distichlis* vegetation re-establishes dominance, and *Scirpus* species abundance declines (Suisun Ecological Workgroup 2001). Artificially stabilizing salinities at low levels during the summer and fall by operation of *salinity* control gates would subdue this pattern of climate-driven vegetation fluctuations, and probably reduce suitability and extent of *Cordylanthus mollis* ssp. *mollis* habitat in Suisun Marsh.

Reduced *freshwater* outflows caused by dams and diversions on the Sacramento and San Joaquin rivers could induce artificially high *salinity* in otherwise *brackish marsh* soils, causing declines in growth and reproduction in *Cordylanthus mollis* ssp. *mollis* (U.S. Fish and Wildlife Service 1997a). This hypothesis, however, is not supported by field evidence that indicates *Cordylanthus mollis* ssp. *mollis* is often particularly abundant in the more saline portions of the *brackish tidal* marshes in which it occurs (salt *pan* edges, high *marsh* zone). It occurs in true salt *marsh* rather than *brackish marsh* (Point Pinole; historically also near San Rafael). Based on observation of the widespread decline in *population* sizes throughout its range in the late 1990s following several years of above-average rainfall (B. Grewell pers. comm. 1997-1998; P. Baye pers. observ. 1997-1998), it appears more likely that long-term *tidal marsh* freshening or dampening of *salinity* variation, rather than progressive salinization, are the greater threats to the species.

In a 2004 survey at Rush Ranch, direct destruction of *Cordylanthus mollis* ssp. *mollis* habitat by feral hogs was frequently encountered (Grewell 2004). Feral hogs were observed rooting and overturning vegetation in former *Cordylanthus mollis* ssp. *mollis* *population* sites. Although Solano Land Trust has permitted limited hunting of feral hogs during closed hours of the reserve, the threat to habitat remains. Hazardous waste remediation at Middle Point (U.S. Navy) resulted in partial destruction of a *Cordylanthus mollis* ssp. *mollis* *population* prior to the species listing as endangered (Ruygt 1994).

Other potential threats to *Cordylanthus mollis* ssp. *mollis* include spills of crude oil or refined petroleum products. Crude oil spills tend to deposit near the high *tide* line where the species is most abundant. Oil spills could have adverse effects on seedling emergence if they occur in winter-spring, and could injure flowering *populations* in summer. In the event of an oil spill, cleanup activities would be concentrated in the high *marsh* zone. Oiling or raking for removal of oiled debris could adversely affect soil *seed banks* of *Cordylanthus mollis* ssp. *mollis*, impairing its regeneration. More volatile refined petroleum products, such as gasoline, with greater potential to penetrate into *marsh sediments*, may require *sediment* removal for remediation. For example, a gasoline line leak on October 4, 2000 contaminated *brackish tidal marsh* near Bay Point, Contra Costa County, in suitable *Cordylanthus mollis* ssp. *mollis* habitat. Soil or *sediment* removal in *tidal* marshes supporting *Cordylanthus mollis* could cause irreversible damage to

populations and habitat. This is particularly pertinent to *seed banks* that take many decades to accumulate. These are a rich *genetic* reservoir. Avoidance of *populations* during oil response may be difficult or ineffective during the non-flowering seasons.

Public access and recreation trails (e.g., jogging trails, bike trails) are often placed at the edges of *tidal* marshes, and sometimes branch directly through *Cordylanthus mollis* ssp. *mollis* *populations* (e.g., formerly at northeast Southhampton Marsh). Trail disturbances have dual effects on *populations*. Initial disturbance may harm *Cordylanthus mollis* ssp. *mollis*, but may also reduce density and cover of closed *marsh* vegetation and create favorable semi-open conditions suitable for expansion into unoccupied habitat. Chronic or increasing trampling, or progressive expansion of *marsh* footpath networks, however, would cause decline or local extinction of *Cordylanthus mollis* ssp. *mollis*.

Some habitat restoration projects may paradoxically have adverse impacts on *Cordylanthus mollis* ssp. *mollis*, at least in the short term. When *brackish* marshes with limited *tidal* range are restored to full or increased *tidal* action, rapid increase in *tidal* range can cause “drowning” of *populations*. Although *marsh* succession in restored marshes is likely to result in additional habitat for *Cordylanthus mollis* ssp. *mollis* in the long term, there are random factors that may result in failure to recolonize suitable habitat. The risk of recolonization failure would be high if *refugial populations* are not protected and managed in the interim successional phases of restoration.

Wetland regulation policies can have a great impact on habitat and *population* viability of *Cordylanthus mollis* ssp. *mollis*. Development, expansion, or improvement of urban shoreline facilities (e.g., marinas, docks, utility pipes, dredge disposal/re-use facilities, road improvements, or residential/commercial development) can directly eliminate or indirectly degrade suitable habitat or *populations*. The Federal and State permitting processes do not lend themselves to comprehensive impact assessments for rare plants as a result of their project-by-project focus, short timeframes, and limited resources. Federal and state authorization of activities that impact wetlands often assess impacts based on acreage. Assessment often assumes that if *Cordylanthus mollis* ssp. *mollis* individuals lie outside a project “footprint” at the time a survey is conducted, the species will not be adversely affected if habitat acreage loss is minimized. Minimization requirements emphasizing compensation for acreage of direct impacts sometimes do not consider biogeographic context, regional function, or *demographic* importance of the particular sites or *populations* for endangered plants. It is important to realize the influence of indirect impacts to *population* viability, and that small patches of restored new habitat are not equivalent to established areas within larger marshes.

Evaluation of potential impacts to endangered plants, such as *Cordylanthus mollis* ssp. *mollis*, for wetland permit applications is sometimes limited to incomplete surveys, or based on inconclusive short-term negative surveys in the brief permit application and review process. The practice of focusing regulatory review only on sites proven to be occupied by endangered plant species is biased against protection of suitable habitat for long-term conservation. *Population* levels of *annual* plants are not static. Brief survey periods are particularly biased against detection of *annual* plants, such as *Cordylanthus mollis* ssp. *mollis*, with fluctuating *populations* that may fail to emerge from persistent *seed banks* in some years. Wetland regulatory agencies

have sometimes issued authorizations before adequate survey results were conducted, deferring surveys and avoidance requirements to pre-construction surveys after permits were issued. In some cases of regional (general) wetland permits for activities such as *levee* repair or ditching in *tidal* marshes, there have been no adequate rare plant surveys required. Overall, these regulatory practices increase the probability of harming undetected latent *populations* due to false negative surveys for colonies that emerge intermittently.

Wetland regulatory agencies with jurisdiction in the geographic range of *Cordylanthus mollis* ssp. *mollis* have tended to be permissive towards projects with small acreage impacts and low levels of public comment, and have performed limited analysis of cumulative impacts of those projects. This practice is likely to cause progressive losses of suitable habitat for the species, since most *tidal* wetland fill projects are located at the upper landward margins of marshes or along *levees*.

Vineyard expansion in North Bay counties increased rapidly in the 1990s, and vineyard plantings have been attempted in *diked* agricultural baylands within San Pablo Bay. Economic pressures to convert relatively unproductive agricultural land to grape production could foreclose many opportunities to restore *tidal* marsh within the historical range of *Cordylanthus mollis* ssp. *mollis*, and may preclude its recovery in substantial portions of its range.

In the final listing rule, intense seed predation by insects was reportedly observed at Joice Island and Hill Slough within the Suisun Marsh in Solano County (U.S. Fish and Wildlife Service 1997a). Insect predation reportedly was responsible for decline in one of the largest *populations* of *Cordylanthus mollis* ssp. *mollis*. Since the time of listing, much light has been shed on the specifics of *C. mollis* ssp. *mollis* seed predation which may pose a threat to *populations* in Suisun Marsh.

Cordylanthus mollis ssp. *mollis* seed production can be significantly influenced by pre-dispersal seed predation from moth larvae (*Saphenista* spp., Tortricidae and salt marsh snout moth, *Lipographis fenestrella*, Pyralidae) (Ruygt 1994; Grewell *et al.* 2003). Areas with muted *tidal* regimes can support the subspecies (California Department of Water Resources 1994), but increased *tidal* muting can constitute a threat to *C. mollis* ssp. *mollis* by increasing the prevalence of unsuitable host plants, and by changing the balance of seed production to seed predation maintained between the plant and seed-eating moths, such as various *Saphenista* species (Grewell 2004, Grewell *in litt.* 2006a). The moth larvae burrow in the *sediment* during part of their life cycle, so reduced *tidal* flooding may improve their survivorship. Under full *tidal* regimes, the interaction between the rare Lepidopteran moth (*Cordylanthus mollis* specialist) and its rare plant host appears to be in balance (Grewell *et al.* 2003, Grewell 2004).

The extent of granivory at Benicia State Recreation Area and Fagan Slough Ecological Reserve were low and these *populations* did not appear to be limited by granivores. However, at sites where *hydrology* was muted, pre-dispersal granivory was extremely high. This has been especially problematic in the Hill Slough area of Suisun Marsh, where the *C. mollis* ssp. *mollis* *population* remains persistent, but under muted *tidal* regimes *population fecundity* has continued to decline (Grewell *in litt.* 2006a).

Sea level rise and associated flood control responses may impose significant long-term threats to conservation of *Cordylanthus mollis* ssp. *mollis*. Conservation of high *marsh* zones requires landward *transgression* (displacement) of the *marsh* profile on broad sloping plains. Many alluvial terraces and valleys adjacent to the *estuary* are bordered by steep *levees* or are already converted to intensive agriculture, residential, or commercial development. In Suisun and northern San Pablo Bay, however, some undeveloped grazing land remains. If rates of sea level rise increase, conflicting needs for flood protection, agriculture, and *marsh transgression* could effectively compress *tidal marsh* zones to a point at which they could not support *Cordylanthus mollis* ssp. *mollis* habitat. Land use planning and economic pressures that favor conversion of “underdeveloped” grazing lands contribute to the loss of potential *transgressive* high *marsh* habitat for long-term viability of the species.

***c. Suaeda californica* (California sea-blite)**

1) Brief Overview

Suaeda californica, California sea-blite, was listed as a federally endangered species over its entire range on December 15, 1994 (U.S. Fish and Wildlife Service 1994) with a recovery priority number of 8, based on a moderate degree of threat, high potential of recovery, and its taxonomic standing as a species (U.S. Fish and Wildlife Service 1983). It is not listed as endangered or threatened by the State of California. Naturally-occurring *Suaeda californica* is now restricted to the southernmost area of its historical range on the shorelines of Morro Bay, where it grows on sandy salt *marsh* edges and high *tide* lines of sheltered estuarine beaches. Numerous threats, both natural and human-caused, exist and are exacerbated by the very low number of individuals, restricted geographic range, and narrow habitat requirements.

2) Description and Taxonomy

Description. *Suaeda californica* S. Watson (California sea-blite, **Figure II-5**) is a salt-tolerant (*halophytic*) member of the Chenopodiaceae (goosefoot family). It grows as a spreading or mounding *subshrub*, woody only at the base. It is usually about 60 centimeters (2 feet) in height, but sometimes reaches over 80 centimeters (3 feet), and spreads up to about 200 centimeters (6 to 7 feet) in width. Individual plants do not appear to form clonal colonies. Leaves are generally pale to *glaucous* green, densely crowded and overlapping, nearly lacking a leafstalk, narrow to nearly needle-like, and up to 3.5 centimeters (nearly 1.5 inches) long. Flowers are not confined to the ends of branches, but occur in scattered clusters of one to three (rarely up to five) at the base of leaves. Flowers are radial, 2 to 3 mm (about 0.1 inch) in diameter, and are either perfect (both pollen- and seed-bearing) or carpellate (seed-bearing only). When flowers occur in clusters of three, the terminal flower is typically perfect and the lateral ones smaller and carpellate. There are five protruding *stamens*, and a cone-shaped ovary with three *stigmas*. The *calyx* lobes are glabrous (hairless) and rounded, or hooded (Munz 1959, Ferren and Whitmore 1983, Ferren 1993).

Taxonomy. *Suaeda californica* was first described by Sereno Watson in 1874, based on type material collected by Bolander and Kellogg in San Francisco Bay salt marshes. Amos Heller published the name *Dondia californica* in 1898, recognizing the genus name used by Michel Adanson in 1763. However, the name *Suaeda* has been conserved (Abrams 1944). Munz (1959) recognized several previously recognized taxa as subspecies of *Suaeda californica*, and described the range as extending from San Francisco Bay south to Lower (Baja) California. Ferren and Whitmore (1983) noted that much of what had been identified as *Suaeda californica* in southern California was a distinct *taxon*, which they named *Suaeda esteroa*. Further study revealed that the only extant *populations* of *Suaeda* that resemble the type specimen of *Suaeda californica* are those that occur in the vicinity of Morro Bay. In his revision of the genus, Ferren (1993) recognized *Suaeda californica* as a full species.

The previous taxonomic ambiguity of the genus in California has resulted in confusion in reports of the geographical ranges of *Suaeda* taxa on the California coast (Fisher *et al.* 1997, Ferren and Whitmore 1983). Even herbarium collections contain some misidentified specimens. *Suaeda taxifolia*, woolly sea-blite of the southern California coast, has been treated by some authors as varieties of *Suaeda californica* (vars. *pubescens* Jeps. and *taxifolia* [Standl.] Munz). *Suaeda taxifolia*, in addition to morphological distinctions (pear-shaped ovary, dense hairiness), typically colonizes coastal bluffs as well as salt marshes. In contrast, most collections of *Suaeda californica* are from salt *marsh* edges or estuarine beaches; it is rarely reported from bluffs at elevations much above sea level.



FIGURE II-5. *Suaeda californica* (Photo Credit Valary Bloom, USFWS)

Many reports of *Suaeda californica* from southern California are erroneous due to confusion with *S. esteroa* (Ferren and Whitmore 1983). *Suaeda esteroa* is restricted to estuaries of

southern California (south of Point Conception) and Baja California. It is ecologically similar to *Suaeda californica*, but can be distinguished by a number of morphological traits.

Several species found within the overall geographic range of *Suaeda californica* may be casually misidentified as it. The most similar is *Suaeda moquinii* (alkali blite or bush seepweed), which is generally found in the Great Valley. In the San Francisco Bay area, *Suaeda moquinii* has historically been restricted to saline or *alkaline seasonal wetlands*. Most *populations* of *Suaeda moquinii* in the San Francisco Bay area are from inland, non-*tidal* localities, but near Fremont and Milpitas it occurs in non-*tidal alkaline/subsaline* wetlands very close to the bay, even in some *diked* historical baylands. *Suaeda moquinii* is generally absent in *tidal* shorelines where *Suaeda californica* would occur, and there are no valid historical records of *Suaeda californica* known from southeastern San Francisco Bay. *Suaeda moquinii* is distinguished from *Suaeda californica* by its open *inflorescences* of flowers clustered at upper ends of stems only, smooth leaf-scars, and widely spaced, non-overlapping (to slightly overlapping) leaves and leaf-like bracts. It also has a pear-shaped ovary. Nonetheless, specimens of *Suaeda moquinii* collected from San Francisco Bay area localities have occasionally been erroneously identified as *Suaeda californica*.

3) Population Trends and Distribution

Historical distribution. *Suaeda californica* was originally reported to range from San Francisco Bay to southern California because of past taxonomic confusion with *Suaeda taxifolia* and *Suaeda esteroa*. As the *taxon* is now narrowly interpreted, the historical range of *Suaeda californica* was limited to the San Francisco Bay Estuary south to Morro Bay. In recent ecological time (latter part of the *Holocene* epoch), its distribution was probably *disjunct* with few, if any, plants between the two *population* centers around San Francisco Bay and Morro Bay.

Based on historical accounts and herbarium collections, it appears that the distribution of *Suaeda californica* was concentrated in the central part of the San Francisco Bay Estuary, with most collections from the Oakland-Alameda area. It was sparsely distributed from approximately Point San Pablo, Contra Costa County (“San Pablo Landing” of Jepson 1911), to San Leandro, Alameda County, and San Francisco County. One *disjunct* collection is known from Palo Alto (Santa Clara County). Assuming correct identifications, early reports suggest that *Suaeda californica* was an infrequent component of the salt *marsh* vegetation of San Francisco Bay.

Brandege (1892) described the distribution of *Suaeda* in San Francisco County from two localities, south San Francisco (the southeastern portion of San Francisco south of Hunters Point; Howell *et al.* 1958) and Visitacion Bay, both along the city’s east shore near the San Mateo County border. These locations appear on early U.S. Coast Survey maps as pocket salt marshes in drowned valleys between headlands associated with narrow *beach ridges* derived from coarse *sediments* such as sand or *shell hash* (Greene 1894, Jepson 1911). The same early topographic maps depict in detail another larger sand spit and *backbarrier* salt *marsh* that occurred in the Presidio. No known reports of *Suaeda californica* exist from the Presidio Marsh.

Best *et al.* (1996) cite a putative record of *Suaeda californica* (as “*Dondia California*” [*sic*]) near the Petaluma River based on the remnant plant content of local adobe bricks dating from the 1830s and 1840s (Hendry and Kelley 1925); however, the accuracy of this identification is highly doubtful because no other *Suaeda* species are reported in the floras of Marin or Sonoma counties (Howell 1949, Best *et al.* 1996), and this area lacks salt *marsh* habitat. Regardless, the dubious adobe brick report is repeated in plant databases (California Natural Diversity Database 1997, CalFlora 2000, California Native Plant Society 2008).

No valid reports or collections of *Suaeda californica* from San Francisco Bay have occurred since the mid-twentieth century. Despite extensive surveys (P. Baye unpubl. data 1991-1999), the last confirmed historical occurrence was a 1958 collection in San Leandro.

Current distribution. Until 1999, *Suaeda californica* was considered extant in Morro Bay, but *extirpated* at its *type locality*, San Francisco Bay. It now is known from five locations in the Morro Bay area as well as at four known reintroduced locations in San Francisco Bay: Pier 98 (Heron’s Head Marsh), Pier 94, Emeryville Crescent, and Robert’s Landing.

Suaeda californica has a brief history of *reintroduction* to San Francisco Bay. Two pilot projects were implemented in 1999 at Crissy Field (National Park Service) and Pier 98 (Port of San Francisco) using clonal stock originating from Morro Bay plants (Baye 2006). Both reintroduced *populations* failed; Crissy Field failed because of impaired *tidal hydrology* and the Pier 98 *population* declined because of unsuitable substrate (Baye 2006). The Pier 98 *reintroduction*, however, resulted in several years of seed reproduction and apparent natural recruitment of a small *population* of highly vigorous *Suaeda californica* on the thin *shell hash* (fine oyster shell fragments) *beach ridges* along an adjacent unrestored urban shoreline. In 2003, the *population* comprised 20 mature plants, producing many tens of thousands of seeds (Baye 2006).

In 2006 the Port of San Francisco and Golden Gate Audubon Society initiated a local *reintroduction* of *Suaeda californica* to a reconstructed sand beach *ecotone* along a small urban salt *marsh* at Pier 94, San Francisco (Baye 2006). The *founder population* was grown from seed collected at Pier 98. At last monitoring all individuals were surviving and growing rapidly.

In 2007, 14 transplants of *Suaeda californica* were introduced along the high *tide* line of East Bay Regional Park’s Eastshore State Park near Emeryville Crescent, Alameda County, in coordination with the U.S. Fish and Wildlife Service and its contractor. Though four transplants died rather quickly due to low rainfall, the remainder were thriving and many were observed flowering at last monitoring (P. Baye pers. comm. 2007). An additional *reintroduction* of eight plants at Robert’s Landing Marsh, Alameda County, was conducted in 2008 (Bloom pers. observ. 2008); however, it is too soon to know if this *population* will be self-sustaining. This site is owned and managed by the City of San Leandro.

In Morro Bay, *Suaeda californica* occurs along the salt *marsh* edges, estuarine beaches, and low bluffs and *scarps* along the shoreline of Morro Bay, San Luis Obispo County and also at the mouths of Old and Villa Creeks and the bluffs at San Geronimo Creek near Cayucos, north of Morro Bay (**Figure II-6 and Figure II-7**). The species distribution was mapped after

comprehensive field surveys of Morro Bay in 1992 (Hillaker 1992), and resurveyed by the U.S. Fish and Wildlife Service from 1997 to 2000 (P. Baye unpubl. data 2000). Several factors indicate that the numerous colonies in Morro Bay constitute a single *population*: (1) Morro Bay is a natural *hydrologic* unit for seed dispersal, comprising a sheltered, enclosed *embayment*; (2) Morro Bay is separated from similar sheltered *embayments*; and (3) the nearest suitable habitats of significant size are occupied by other species of *Suaeda*.

Morro Bay subpopulations of *Suaeda californica* include:

- (1) North Shore: vicinity of Morro Bay State Park and the *tidal* inlet throat (Morro Channel, Morro Bay harbor; heron rookery, Fairmount Point);
- (2) Sand spit *backbarrier* shoreline: Morro Dunes Nature Preserve;
- (3) Southeast Morro Bay: Sweet Springs Nature Preserve and Los Osos/Cuesta-by-the Sea shoreline;
- (4) Baywood Park bluffs vicinity: pocket marshes and low bluffs in ancient dunes, near end of Santa Ysabel Avenue; and
- (5) Cayucos: mouths of Old and Villa Creeks and bluffs at San Geronimo Creek.

Small colonies have been identified along the urbanized Embarcadero shoreline of Morro Bay *tidal* inlet, approximately between the end of Morro Bay Boulevard and the Morro Bay boat launch (Hillaker 1992). Large gaps in distribution exist at the tip of the sand spit where unstable mobile dunes migrate directly into the bay and along the dredge disposal site at the extreme north end. The species is largely absent along the high *marsh* shoreline of the *marsh* deltas of Chorro and Los Osos Creeks. It occurs only at the northwest corner of the Chorro Creek delta *marsh*, at Morro Bay State Park.

Suaeda californica in Morro Bay declined dramatically during the late 1990s when it was reported along nearly all of the shoreline, with colonies growing often continuously from the southern end of the bay to the northern reaches of the sand spit. It was absent only along shoreline segments with highly mobile unvegetated dunes (Hillaker 1992, P. Baye unpubl. data 1997). The severe winter storms of 1997-98 scoured away all but small remnants of this formerly extensive *colony*. The *driftlines* in eroded gaps briefly supported a flush of *Suaeda californica* seedlings in 1998, but few survived by the spring of 1999 (P. Baye unpubl. data). Relatively sheltered, smaller *Suaeda californica* colonies in the northern part of the bay were less heavily impacted by erosion, and were relatively intact. Between 1998 and 2000, nearly all known occupied habitat of *Suaeda californica* in Morro Bay was resurveyed after the mass dieback of the 1998 flush of seedlings (Baye pers. comm. 2004). The total Morro Bay *population* size of the species was estimated to be nearly 360 mature plants in 2000.

There are no reported field estimates of the total *Suaeda californica* *population* prior to the 1997-1998 storms when most of the plants along the spit shoreline grew in continuous colonies, not as discrete identifiable individuals. However, a conservative estimate suggests that the spit

subpopulation alone probably supported at least 1,700 to 2,400 plants (Baye pers. comm. 2004). This estimate may be low because of the irregular shoreline and the presence of smaller plants mixed in colonies.

In 2002, California Department of Parks and Recreation initiated a *reintroduction* project aimed at restoring *Suaeda californica* habitat through removal of *non-native* vegetation along the *estuary* edge. The expansion of the known range of *Suaeda californica* was also augmented by propagation, then introduction. A *population* census of the *reintroduction* areas was conducted in December 2004 (California Department of Fish and Game 2006) where *population* estimates using two different methods, ranged from 2,934 to 3,597 individuals. Restoration resulted in an expanded *population* at one site at the North Shore *subpopulation*. The exotics removal work at Morro Estuary Natural Preserve allowed the *population* to naturally expand and persist in 2005 as reproductive plants.

Herbarium records indicate occasional historical occurrences of *Suaeda californica* outside of Morro Bay in the vicinity of creek mouths (Hardham 2710, 1957) and coastal bluffs (R. Ferris, 1929, DS206274) near Cayucos. California Department of Parks and Recreation's 2005 survey of *Suaeda californica* in and around Morro Bay revealed a total of 28 to 30 plants surviving near Cayucos at the mouths of Old Creek and Villa Creek and on clay soils on coastal bluffs at San Geronimo Creek (California Department of Fish and Game 2006). These could be significant *populations* because of their isolation and the environmental extremes to which they are adapted.

4) *Life History and Ecology*

Suaeda californica produces seeds throughout its lifespan. Reproduction appears to be entirely by seed (sexual); there are no known reports of natural regeneration from vegetative fragments. The spread of individual plants can be extensive, and sometimes resembles clonal *populations*. However, they have not been observed to spread clonally. Vegetative stem cuttings of *Suaeda californica* treated with synthetic auxins (hormones) are easily rooted for artificial propagation (P. Baye pers. observ. 1991-1999).

Reproductive maturity may in some cases be reached in as little as one year (P. Baye unpubl. data 1998). Flowering occurs on portions of the current year's shoot growth, usually on lateral branches of older wood. Flowers typically appear from May to October, but mostly in late summer. Occasional flowers may be found at other times of the year, sometimes emerging as early as late spring (McMinn 1939, Baye pers. observ.). Differences in flowering phenology may be an indication of *genetic* variation. One entire *colony* of *Suaeda californica* on Pickleweed Island, Morro Bay, was observed to flower precociously in April, while adjacent plants and all other colonies were entirely vegetative (P. Baye unpubl. data 2000). The longevity of individual plants is unknown, but large woody plants in stable substrate appear to live for over a decade.

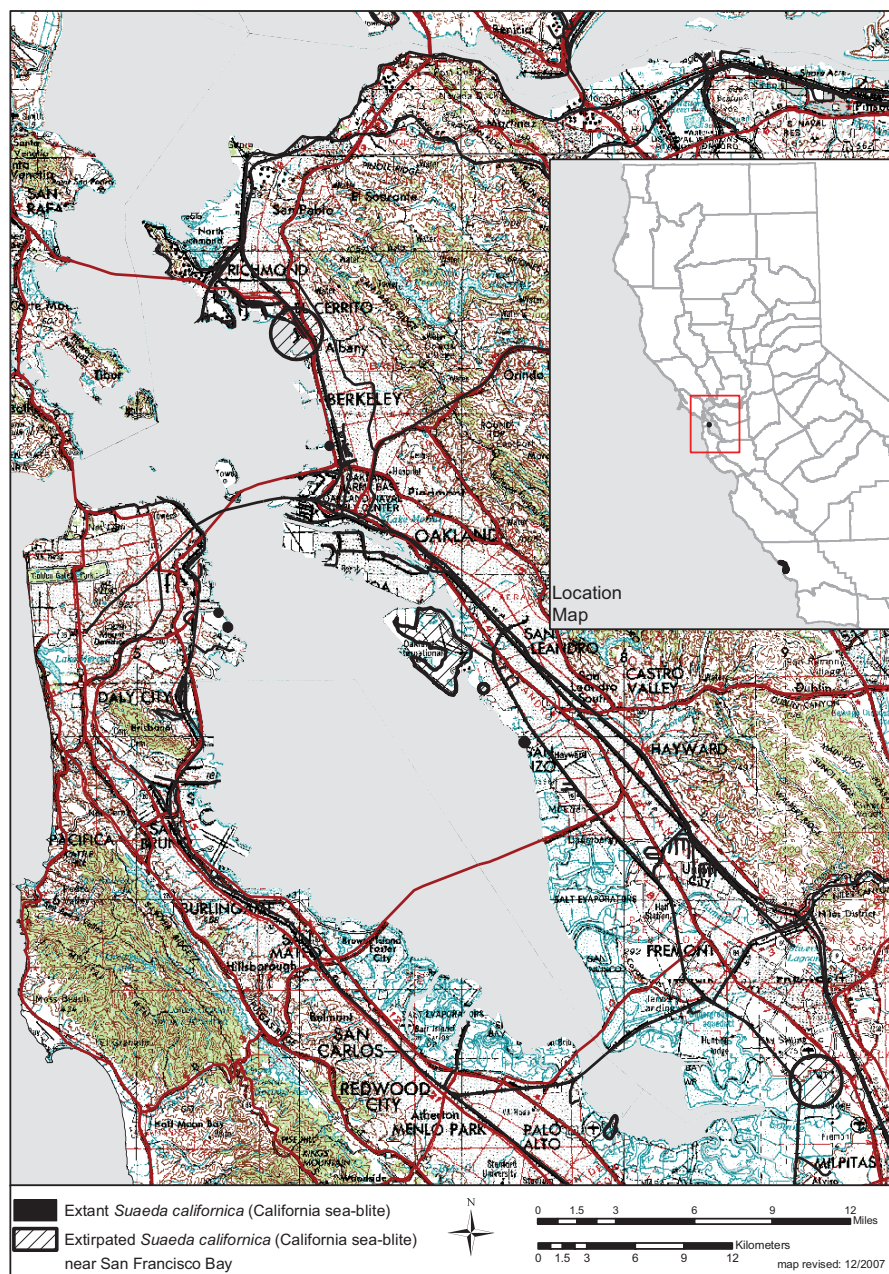


Figure II-6.

Figure II-6. Distribution of *Suaeda californica* in San Francisco Bay



Figure II-7.

Figure II-7. Distribution of *Suaeda californica* in Morro Bay

Very little information is available on the breeding system of *Suaeda californica*; however, a predominantly outcrossing breeding system would be expected for this wind-pollinated, often colonial, shrub. Abundant seed (many hundreds per plant) is produced on fruiting plants at Morro Bay. The ability of isolated plants in cultivation to produce seed (P. Baye pers. observ. 1998) suggests that at least some individuals possess a degree of self-compatibility. Abundant seed set occurred spontaneously in outdoor container-grown nursery plants at the Golden Gate National Recreation Area nursery in San Francisco in 1998. These seeds were *viable* and produced vigorous seedlings (E. Heimbinder pers. comm. 1999).

Based on observations by *marsh* ecologist Peter Baye, abundant seedling establishment at Morro Bay appears to be episodic, corresponding to storm events that cause both vegetation gaps and deposits of driftline debris with seeds. Seedlings were widespread and abundant along the *backbarrier* shoreline following the erosive winter storms of 1998. Many thousands of seedlings and multiple-branched juvenile plants had established in the erosion zone in *driftlines* and litter rafts by late April 1998. Seedlings rooted in debris rafts without roots in the *marsh* substrate were subject to high mortality. No evidence of long distance dispersal and colonization was observed. Re-surveys of the extensive 1998 seedling colonies in April of 1999 and 2000 revealed only regeneration of remnant mature shrubs that survived erosion. No juvenile or young mature plants were detected, indicating extremely high mortality of the post-storm cohort of seedlings. In contrast, the colonies of mature *Suaeda californica* at the north end of Morro Bay were mostly unaffected by the 1998 storm. These narrow, dense colonies acted as a significant *refugia* for survival and seed production during the *catastrophic* mortality that affected most of the *population* along the bayshore of the central sand spit.

5) *Habitat Characteristics/Ecosystem*

Suaeda californica is largely restricted to the narrow high salt *marsh* zone in Morro Bay, often within the *wrack* line of storm *tides*. In Morro Bay this habitat occurs mostly on sandy substrates or pure sand, such as *scarps* in ancient *Pleistocene* dunes (Morro Channel and harbor shore, Baywood Park, Los Osos), modern dunes and estuarine beaches (Morro Spit), and small low *spits* and *marsh berms* (Sweet Springs Marsh, Pickleweed Island). *Suaeda californica* also occurs among rocks placed over sandy fill material in artificial shorelines (northern Morro Bay sites), and on well-drained sandy *marsh* peat at the edge of eroding *marsh scarps* (Baywood Park near the end of St. Ysabel Street). *Suaeda californica* exists on *shell hash* beach ridges in San Francisco. The most environmentally extreme habitat occupied by *Suaeda californica* is the guano-enriched bluffs below the cormorant and heron rookery along Morro Channel. Intensive local deposition of urea- and ammonia-containing guano has killed eucalyptus trees and all terrestrial vegetation except *Suaeda californica*, which develops luxuriant growth with rich blue-green plants many meters across that grow from the base of the bluff upwards. This situation well illustrates the affinity of specialized *Chenopodiaceae* species for extreme levels of soil sodium or nitrogen (Waisel 1972).

Suaeda californica habitat, estuarine sand beaches within salt *marsh*, is very scarce in San Francisco Bay Estuary today. There is currently only one remnant historical sandy salt *marsh* in the *estuary* (Whittell Marsh, Point Pinole), and a few small pockets of recently formed sandy salt marshes where artificial fill has eroded and been redeposited (*e.g.*, Albany dump shoreline,

frontage road along I-80 south of Albany; portions of the southeastern San Francisco shoreline). Only one sand spit has naturally reformed along the San Leandro shoreline within the last two decades (Robert's Landing, San Lorenzo Creek mouth). Bayward edges of salt marshes and levees in the vicinity of Redwood City and Palo Alto today still develop low ecotonal *beach ridges* about 0.5 meter (1 to 2 feet) above the *marsh* plain composed of *shell hash*. These *beach ridges* become *marsh berms* covered with high salt *marsh* vegetation, including native species such as *Grindelia stricta* var. *angustifolia* (gumplant), *Frankenia salina* (alkali-heath), and *Sarcocornia pacifica* (pickleweed). These *shell hash beach ridges* probably provided habitat for *Suaeda californica* in this portion of the bay where sand supplies were minimal.

High rates of sand deposition and erosion limit growth and survival of *Suaeda californica* at Morro Bay. Colonies occur occasionally at the advancing edges of dunes where they grow through and at least temporarily keep pace with sand *accretion*. At some locations along the bay shoreline, vigorous growth persists at elevations up to 1.2 meters (4 feet) above the high salt *marsh* surface. More often, however, rapidly moving dunes bury and kill *Suaeda* colonies and other *marsh*-edge vegetation in their path.

A significant source of nutrients to *Suaeda californica* colonies at Morro Bay is provided by thick wracks of decomposing *Zostera marina* (eelgrass), which form *driftlines* in the zone of highest *tidal* influence. This provides nitrogen for plants growing in nutrient-deficient sand. Colonies that grow in local conditions that discourage deposition or retention of *Zostera driftlines* (e.g., steep artificial rock slope of the marina, low bluffs in ancient dunes) often have sparse yellowish gray-green foliage, compared with the luxuriant grayish blue-green foliage of plants in *driftlines* and guano-enriched sandy soils at the heron rookery (P. Baye unpubl. data 1997-2000). Other major mineral nutrients (particularly potassium and calcium) are presumably provided by seawater.

The salt tolerance of *Suaeda californica* has not been evaluated experimentally, but limited field evidence suggests that subsurface flow of groundwater from adjacent dunes may reduce *salinity* of the root zone in many situations (P. Baye unpubl. data 1999).

Although described as a species of salt *marsh* habitats (Munz 1959, Hickman 1993, Sawyer and Keeler-Wolf 1995), *Suaeda californica* occurs only in a narrow *ecotone* between the extensive middle salt *marsh* zone characterized by *decumbent* to prostrate *Sarcocornia pacifica*, *Triglochin concinna* (creeping arrow-grass), and *Jaumea carnosa* (fleshy jaumea)); and below upland vegetation at the edge of the *marsh*, often dominated by stable dune scrub with *Eriophyllum staechadifolium* (woolly sunflower) or *non-native* *Carpobrotus edulis* (iceplant or hottentot-fig) and hybrids. This high *marsh ecotone* in Morro Bay is typically only about 1 to 2 meters (3 to 7 feet) wide depending on slope. Within this zone, *Suaeda californica* associates with *Distichlis spicata* (saltgrass), *Atriplex watsonii* (Watson's saltbush), *Atriplex triangularis* (spearscale), *Atriplex californica* (California saltbush), *Sarcocornia pacifica*, *Frankenia salina*, and *Jaumea carnosa*. *Isocoma veneta* ssp. *vernonioides* also occurs locally in the high *marsh* zone with *Suaeda californica*. Occasionally, *Cuscuta salina*, a parasitic dodder, occurs on *Suaeda californica* in this zone, but no lasting injury has been observed (P. Baye pers. observ.). *Suaeda californica* also occurs adjacent to colonies of the rare *Cordylanthus maritimus* ssp. *maritimus* (salt marsh bird's beak) and *Lasthenia glabrata* ssp. *coulteri* (Coulter's goldfields) in high salt

marsh at Sweet Springs Marsh. *Suaeda californica* is probably associated with the corresponding northern varieties/subspecies of these taxa in San Francisco Bay. Native dune plant species include *Ericameria ericoide* (mock-heather), *Croton californicus* (California croton), *Senecio blochmaniae* (Blochman's leafy-daisy), *Amsinckia spectabilis* (coast fiddleneck), and *Achillea millefolium* (yarrow). The most frequent and important *non-native* species that associate with *Suaeda californica* are *Carpobrotus edulis* and hybrids with *Carpobrotus chilense*. Competition with creeping, mat-forming *Carpobrotus* is evident where it extends down from dunes and through, up, and over narrow colonies of *Suaeda californica*.

6) Critical Habitat

Critical habitat has not been designated for *Suaeda californica* (U.S. Fish and Wildlife Service 1994).

7) Reasons for Decline and Threats to Survival

Most species covered in this draft recovery plan are threatened by similar factors because they occupy the same *tidal marsh* ecosystem. These general threats, faced by all covered species, are discussed in greater detail in the Introduction section of this draft recovery plan (section I.D.). Specific threats to *Suaeda californica* are described below.

There are numerous threats to the survival of *Suaeda californica*. The impacts of these threats are intensified by the very restricted geographic range and extremely narrow ecological distribution of this species.

Alteration and Loss of Habitat

The historical rarity of *Suaeda californica* in San Francisco Bay may have been due in part to the natural rarity of its sandy high *marsh* and beach habitat, but its extirpation seems related to the early spread of urban and port development over the East Bay shoreline from Richmond to Alameda, centered around Oakland (P. Baye pers. comm. 2004). This heavily urbanized area was the center of both the bay's sandy shorelines and *S. californica* distribution. Oakland and Alameda Marshes were filled and urbanized before the 20th century, eliminating *populations* there, but it was the destruction of Bay Farm Island for the construction of the Oakland International Airport in the 1950s and 1960s that probably destroyed the only remaining *viable* core *population* in San Francisco Bay. Other species with affinity for sandy salt *marsh* edges, such as *Atriplex californica*, were also described as occurring either along sandy beaches or sandy *marsh* edges within San Francisco Bay (Brewer *et al.* 1880, Jepson 1911, Greene 1894). These, too, have become *extirpated*.

The Morro Bay *population* has suffered little habitat loss compared with San Francisco Bay, and has relatively abundant habitat there, despite declines following El Niño winter storm erosion. However, it is subject to strong fluctuations in abundance due to natural disturbances, particularly dune migration and shoreline erosion, and its regeneration following disturbance is vulnerable to numerous threats. Though the *population* has in the past been threatened by strong residential and commercial real estate development pressures on the east shore of Morro Bay, centered at Baywood Park and Los Osos, these pressures have been reduced drastically (J.

Vanderweir pers. comm. 2009). Loss of habitat and individuals, and failed regeneration after natural *catastrophes* could cause extirpation of this *population*. Other threats include interference by *non-native* vegetation, trampling, oil spills, sea level rise associated with climate change, excessive dune mobilization, and alteration of shoreline dynamics due to stabilization and shoreline repair projects.

Recruitment failure

Trampling of seedlings in Morro Bay may contribute to the failure of *Suaeda californica* regeneration following *catastrophic* shoreline erosion caused by major storms. Trampling results from both recreational activities (hiking) and by black-tail deer (*Odocoileus hemionus*) *populations* on the sand spit and represents a relatively infrequent threat. However since seedling recruitment is episodic and local, impacts to seedlings (which are difficult to detect) could be severely detrimental at times.. This is indicated by tracks and footprints along the Morro Bay shoreline in a devegetated zone nearly 0.5 meter (less than 2 feet) wide (P. Baye pers. observ. 1997-1999). As recreational pressure on the Morro Bay shoreline increases with local residential *population* and increased visitor use at Montaña de Oro and Morro Bay State Parks, this impact is likely to become more severe.

Competition with non-native species

Exotic *invasive* vegetation, primarily *Carpobrotus edulis* X *chilensis* hybrids (iceplant), *Eucalyptus globulus* (blue gum), and *Cupressus macrocarpa* (Monterey cypress; a native to the Monterey peninsula only), cause significant damage to *Suaeda californica* by direct interference and indirect adverse habitat modification. *Carpobrotus edulis* establishes clonal colonies in adjacent *uplands* above saline influence, and can encroach by transporting nonsaline soil moisture from portions of the clone above the high *tide* line (P. Baye unpubl. data 1997). Most stands of *Suaeda californica* along the perimeter road to Morro Beach State Park have been partially smothered by *Carpobrotus edulis*, which grows through and over the *Suaeda californica* colonies there. *Carpobrotus edulis* impacts are particularly significant for seedling regeneration along the *backbarrier* shore of Morro Bay spit. As the sandy *backbarrier* shoreline retreats into dense continuous stands of *Carpobrotus edulis* on the dunes, *Carpobrotus edulis* overhangs the erosional scarp and forms a canopy that drapes over the base of the scarp and upper shoreline. This sharply reduces or eliminates open seedling habitat for *Suaeda californica*—its regeneration niche. It may also inhibit regeneration of storm-eroded remnants of *Suaeda californica*. Therefore, spread of *Carpobrotus edulis* along the dunes of the *backbarrier* shoreline is likely to reduce *population resilience* of *Suaeda californica*. In fact, removal of *Carpobrotus edulis* near *Suaeda californica* populations has had a striking effect of recovery of the later (Baye, *in litt.* 2009).

Heavy leaf litter and canopy shade from *non-native* trees, *Cupressus macrocarpa* and *Eucalyptus globulus*, are detrimental to seedling habitats for *Suaeda californica*, and apparently cause decline in vigor of remnant stands of mature plants (e.g., near the entrance of Morro Bay State Park and in Baywood Park). Degradation of the ecological niche for seedling regeneration is probably a more severe long-term threat to the viability of the *Suaeda californica population* than local disturbance of existing mature colonies. Stands of *Suaeda californica* have been damaged directly by broken and fallen limbs of *Eucalyptus globulus* adjacent to Morro Beach State Park (P. Baye unpubl. data 1997-2000).

The persistence of suitable and restorable habitat for *reintroduction* of *Suaeda californica* to San Francisco Bay is also threatened by *non-native* vegetation. In San Francisco Bay, the spread of *non-native* *Spartina alterniflora* (smooth cordgrass) and its hybrids with the native *Spartina foliosa* (Daehler and Strong 1996) has caused the conversion of open *mudflat* into stabilized salt *marsh* that traps *sediment* and moderates estuarine wave energy. By spring 2000, nearly one half the length of Roberts Landing Spit, the largest undeveloped sand spit remaining in the bay, and most of the adjacent San Lorenzo Creek delta, San Leandro, were stabilized by *Spartina alterniflora*. This *invasive* vegetation intercepts alongshore transport of sand in the middle and lower *intertidal* zone, and inhibits the wave deposition of the sandy higher elevation *marsh-beach ecotone* that is essential for establishment of *Suaeda californica*. Further spread of the *Spartina alterniflora* hybrid swarm could significantly diminish successful re-establishment of sandy *marsh-beach ecotones*, and preclude the long-term viability of *Suaeda californica* *reintroduction*.

Dredging

Navigational dredging may threaten stands of *Suaeda californica* that have colonized the marina shoreline at Morro Bay State Park. Dredging of the inlet channel steepens the *subtidal* shore profile, probably resulting in shoreline erosion along the unarmored eroding south shore of the interior shoreline of the marina, which could threaten the *Suaeda californica* colony there. The marina *subpopulation* of *Suaeda californica* is particularly significant to the species' conservation because it is highly sheltered from storm wave erosion that threatens the main *population* along the spit's *backbarrier* shoreline.

Predation

In the absence of natural predators, hunting, or management in Morro Bay, deer *populations* are likely to forage intensively along the *backbarrier* shoreline where seeps provide fresh water, soft herbaceous vegetation, and flat travel *corridors*.

Small number of populations

Suaeda californica is vulnerable to extinction in the wild largely because it has been reduced to a very small number of *populations* distributed in a very narrow zone of the Morro Bay and San Francisco Bay shorelines. In Morro Bay, most of the colonies occur along the erodible *backbarrier* shore of the Morro Bay sand spit, which is susceptible to erosion by occasional extreme storm *tides* and high wind-generated waves, and rapid burial by migrating dunes. Severe storm erosion occurred along this shoreline in the winter of 1997-1998, creating an extensive erosional scarp in the narrow *Suaeda* zone. The *population* has not yet rebounded from this event. Although this was a natural *catastrophe* and rebound may occur in time, erosion events may become a recurrent threat if climate change increases storm intensity, frequency, and sea level rise rates.

Climate change

Extreme fluctuations of climate (winter storms, high winds, summer drought) may be associated with global climate change. A series of severe winter storms followed by years of drought could cause *catastrophic* reproductive failure of the species. Global warming and associated sea level rise may also cause long-term changes in the stability of sand beach and dune shorelines (SCOR

Working Group 1991), such as those of Morro Bay spit. *Suaeda californica* occurs in abundance only where the *backbarrier* shoreline is adjacent to dune scrub vegetation that stabilizes dunes. It is sparse or absent where bare mobile dunes retreat over the *backbarrier* shoreline. Many of the remaining colonies are being encroached on by mobile dunes, and are not expected to survive more than a few years. The formation of new “*marsh coves*” (potential *Suaeda californica* habitat) in the lee of stabilizing dunes may occur in the future, but none are foreseeable now. A combination of shoreline retreat and increased dune movement could significantly reduce the largest *subpopulation* of *Suaeda californica*.

Accelerated sea level rise and shoreline retreat could also force conflicts between natural movement of the *Suaeda californica* zone on the east shore of Morro Bay and landowner needs. Where costly residential developments are threatened by shoreline retreat, response typically involves armoring (structural stabilization) of the shoreline (*e.g.*, revetments, seawalls, rip-rap, etc.). Currently, *Suaeda californica* appears to be able to migrate with the slowly retreating shorelines of eastern Morro Bay (Baywood Park, heron rookery).

Oil spills

Oil spills and clean-up operations may have significant adverse effects on *Suaeda californica* populations at Morro Bay, particularly on seedlings. Spilled oil tends to accumulate near the high *tide* line, the narrow *marsh* zone in which *Suaeda californica* is largely restricted. Oil would probably cause high mortality of seedlings and juvenile plants during years of seedling regeneration by coating and smothering small plants with oil, and possibly by direct toxicity. Oil clean-up operations involving mechanical removal (raking, excavation) of oiled sand would also cause significant disturbance of *Suaeda* habitat. Direct toxic effects of oil on older woody *Suaeda californica* are uncertain, but are probably less damaging than effects of clean-up operations.

Other threats to *Suaeda californica* include factors that preclude its recovery outside of Morro Bay. The main cause of its regional extinction in San Francisco Bay—urbanization of the original natural sandy *marsh* habitats—is irreversible. However, the maintenance of steep *levees* constructed of bay mud along portions of San Francisco Bay prevents re-establishment of potential wave-deposited *marsh berms* or sand *beach ridges* and *spits*, which could provide habitat for *reintroduction*. Historical San Francisco Bay *levee* designs, and application of traditional methods of *levee* repair and maintenance, are major impediments to habitat restoration and *reintroduction* of the species to the only other historical habitat in its natural range. The three sites in San Francisco Bay where *reintroduction* has already occurred were carefully selected as some of the few sites where existing *levees* or the maintenance thereof would not negatively affect the plants. Care was also taken to select sites where recreational activities or other pressures would not threaten possible future *populations*.

d. California Clapper Rail (*Rallus longirostris obsoletus*)

1) Brief Overview

California clapper rails were recognized as endangered by the Federal government and added to the List of Endangered Species on October 13, 1970 (U.S. Fish and Wildlife Service 1970). California clapper rails were added to the State endangered species list on June 27, 1971 (California Department of Fish and Game 2005). It has a recovery priority number of 3C, based on a high degree of threat, a high potential of recovery, and its taxonomic standing as a subspecies. The additional “C” ranking indicates some degree of conflict between the conservation needs of the species and economic development (U.S. Fish and Wildlife Service 1983). The first recovery plan for the species was published November 16, 1984 (U.S. Fish and Wildlife Service 1984). Factors currently impacting rail numbers baywide include predation, contaminants, and habitat loss/alteration/degradation.

2) Description and Taxonomy

The California clapper rail (*Rallus longirostris obsoletus*) belongs to the order Gruiformes, in the family Rallidae, which includes rails, gallinules, and coots. The genus *Rallus* consists primarily of marsh-dwelling birds with short rounded wings, large feet, and long toes. Clapper rails generally inhabit coastal salt or brackish marshes.

Description.—The California clapper rail is one of the largest species of the genus *Rallus*, measuring 32-47 centimeters (13-19 inches) from bill to tail (Ripley 1977; **Figure II-8**). Males generally weigh 300-350 grams (0.66-0.77 pound) and females 248-301 grams (0.55-0.66 pound; Taylor 1996). The clapper rail has a hen-like appearance, with a long slightly decurved orange bill, a rufous breast, black and white barred flanks, and white undertail feathers. Juveniles have a paler bill and darker plumage, with a gray body, black flanks and sides, and indistinct light streaking on flanks and undertail coverts. Downy young are black with dark legs (Eddleman and Conway 1998).

Clapper and Virginia rails are morphologically similar and may co-occur in tidal marshes. Clapper rails are larger than Virginia rails, and lack the gray cheeks characteristic of Virginia rails. In addition, the brown back feathers of clapper rails are edged with gray, while the back plumage of Virginia rails is chestnut colored.

Clapper rail call. Because of their secretive habits, clapper rails are most often detected by their calls; visual detection is infrequent. Clapper rails have a wide variety of calls, although few are commonly heard. All calls are variants on a single note, with differences due to changes in intensity, pitch, note length, and interval between notes. Massey and Zembal (1987) grouped clapper rail vocalizations into eight calls, of which four are commonly heard: clapper, kek, kek-burr, and agitated kek. The clapper is the basic species call, serving as a territory pronouncement and for mutual mate recognition. Both sexes clapper year-round, with daily peaks at dawn and dusk. In central San Francisco Bay, vocal activity by California clapper rails was greatest from November through April (Evens and Page 1983). The clapper call is used as the basis for aural population censuses (Evens and Collins 1992, Collins *et al.* 1994, Evens 2000a); however, time of day, tidal height, and weather conditions all affect the frequency of calling (Zembal and Massey 1987).

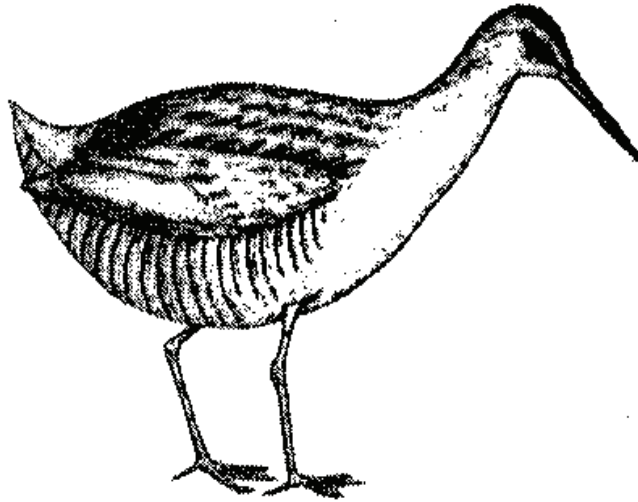


FIGURE II-8. California clapper rail (from California Department of Fish and Game 2000, with permission)

The kek is the second most frequent call, and is confined to the advertisement of non-mated males during the breeding season (Massey and Zembal 1987, Zembal and Massey 1987). The kek-burr is the advertisement of non-mated females, and is only heard during the breeding season. It consists of one or more keks, evenly spaced, usually followed by a burr (Zembal and Massey 1985). Zembal and Massey (1987) suggest that an uneven number of kek calls relative to kek-burrs may represent a skewed sex ratio of rails in a *marsh*. The agitated kek is a response to intrusion or disturbance (Massey and Zembal 1987). Newly hatched chicks emit peeping sounds (U.S. Fish and Wildlife Service unpubl. data).

Taxonomy.—The California clapper rail was first described as a king rail (*Rallus elegans* var. *obsoletus*; Ridgway 1874) until Ridgway (1880) reclassified it as a geographically distinct form of clapper rail. Van Rossem (1929) demonstrated that all Pacific coast *populations* of clapper rails were geographical races of one species, and designated the California race as *Rallus obsoletus obsoletus*. Subsequently, Oberholser (1937) described 25 clapper rail forms as subspecies of the same species, and the California clapper rail became *Rallus longirostris obsoletus*.

Although the taxonomic status of clapper rails is a matter of some debate, the American Ornithologists' Union (1957) distinguishes five subspecies of clapper rails in North America. The California clapper rail is the only subspecies that inhabits the coast of northern California and San Francisco Bay.

3) Population Trends and Distribution

Historical distribution.—California clapper rails were historically abundant in all *tidal* salt and *brackish* marshes in the San Francisco Bay vicinity (Cohen 1895), as well as in all of the larger

*tidalest*uaries from Marin to San Luis Obispo counties. The salt marshes of south San Francisco Bay, including portions of San Mateo, Santa Clara, and Alameda counties, supported the largest *populations* of California clapper rails (Grinnell 1915, DeGroot 1927, Williams 1929, Grinnell and Miller 1944). Gill (1979) identified the Napa River as a North Bay *population* center, which supported approximately 40 percent of the entire *population*.

There are isolated records of rails occurring in urbanized areas of San Francisco (Orr 1939), Oakland, and Berkeley (Lindsdale 1936). Rails were also reported from Point Isabel in Contra Costa County (Williams 1957).

Small *populations* existed in San Pablo Bay along Wildcat Creek/San Pablo Creek in western Contra Costa County (Grinnell and Miller 1944). Newberry (1857) reported clapper rails as very common in the marshes of Petaluma. Bryant (1931) reported rails in Richardson Bay, and an egg set was collected from Corte Madera in 1931 (Gill 1979). In Solano and Sonoma counties, Gill (1979) and Harvey (1980) observed rails at numerous locations in the Napa Marsh complex.

According to survey data, the historical distribution of clapper rails within San Francisco Bay was restricted to marshes west of Suisun Bay. However, systematic survey data from the Suisun Marsh area were not available until the 1970s. Clapper rails have been consistently detected in the Suisun Marsh area since the 1970s, although abundance has been low (Gould 1973, Harvey 1980). It is likely that low numbers of clapper rails were present in this area prior to large-scale *marsh* reclamation.

North of the San Francisco Bay area, clapper rails formerly occurred in Humboldt Bay, Humboldt County (Grinnell and Miller 1944, Gill 1979), and in the Marin-Sonoma *embayments*, which include Bodega Harbor, Tomales Bay, Drakes/Limantour Esteros, and Bolinas Lagoon (Storer 1915, Brooks 1940, Grinnell and Miller 1944). The last record for Humboldt Bay was in 1947 (Wilbur and Tomlinson 1976). There have been several records of clapper rails in Tomales Bay in the late 1990s (J. Evens *in litt.* 2007). Prior to these observations, clapper rails had not been documented in Tomales Bay since 1914, and were presumed *extirpated* as of 1973 (Storer 1915).

South of the San Francisco Bay area, clapper rails formerly occurred in Elkhorn Slough, Monterey County (Silliman 1915), and Morro Bay, San Luis Obispo County (Brooks 1940). Clapper rails were consistently detected in Elkhorn Slough up to 1972, when an estimated 10 pairs were observed (Varoujean 1972). Subsequently, rails were observed only sporadically (Winter and Laymon 1979), and were last documented there in 1980 (Roberson 1985). There are few records of clapper rails in Morro Bay since 1942 (Wilbur and Tomlinson 1976). Despite a 1977 record for Morro Bay (Gill 1979), Harvey (1980) found no evidence of clapper rails there in 1979.

Transient California clapper rails have been occasionally observed at other locations along the coast of California, including the Farallon Islands (Bryant 1888), Pacific Grove (Kimball 1922), Pescadero Marsh (Orr 1942), and Bolinas Lagoon (Harvey 1980).

Current Distribution.—California clapper rails are now restricted almost entirely to the marshes of the San Francisco Bay Estuary where the only known breeding *populations* occur (**Figure II-9 and Figure II-10**). There has not been a recent complete survey of its *population* and distribution within the *estuary*. The California clapper rail *population* was first estimated at 4,200 to 6,000 birds between 1971-1975, of which 55 percent occurred in the South Bay and 38 percent in the Napa Marshes (Gill 1979). Although the *population* was estimated at only 1,500 birds between 1981-1987 (Harvey 1988), the difference between these two estimates is believed to be partially due to survey intensity. Breeding season density data indicate that *populations* remained stable during the 1970s (Gill 1979, Harvey 1980), but reached an estimated all-time historical low of about 500 birds in 1991, with about 300 rails in the South Bay (Harding *et al.* 1998). Rail numbers have rebounded slightly since the early 1990s. However, substantial increases in *population* may be difficult to achieve due to the current *disjunct* distribution of their habitat (Albertson and Evens 2000).

PRBO Conservation Science conducted *estuary*-wide surveys of the San Francisco Bay for California clapper rail between 2005 and 2008. Results of the 2008 survey indicated only 543 rails, compared to 938 rails detected in 2007 (PRBO Conservation Science 2009). In both years, South San Francisco Bay accounted for the majority of rails.

Central/South Bay. The clapper rail *population* in the eastern portion of the South Bay decreased substantially, from 400-500 individuals to 50-60 in 1991-92 (Harvey 1980, U.S. Fish and Wildlife Service unpubl. data), but then rebounded to 330 individuals in 1997-1998. In response to predator management, the total South Bay rail *population* has rebounded since the low of the early 1990s (Harding *et al.* 1998), and was estimated to be approximately 650 to 700 individuals in 1997-1998 (U.S. Fish and Wildlife Service unpubl. data). The highest *population* densities for clapper rails continue to be located in south San Francisco Bay, where clapper rail *populations* presently occur in all of the larger *tidal* marshes. The largest *populations* currently occur in Arrowhead, Dumbarton, Mowry, and Cogswell marshes in the East Bay, and in East Palo Alto and Greco Island in the west bay (Herzog *et al.* 2006). In Alameda County, rails are known to occur in the Emeryville Crescent, Hayward, Old Alameda Creek, Ideal, La Riviere, and Coyote Creek marshes. In San Mateo County, rails currently occur in marshes along Faber/Laumeister, Ravenswood, Seal Slough, and the Colma Creek area. In Santa Clara County, rails occur along Alviso and Charleston Sloughs, and in outboard marshes of Moffett Field and Guadalupe Slough. Clapper rails can also be found in salt marshes fringing the South Bay outboard of salt evaporation pond *levees* and along major *tidal sloughs*.

In 2006, the central San Francisco Bay experienced highest numbers of clapper rails in Corte Madera (Heerdt) and Muzzi Marshes in Marin County (Herzog *et al.* 2006). Other occupied areas include Wildcat Marsh and Oakland Inner Harbor in southern Contra Costa County and Richardson Bay and Creekside Marsh in Marin County (Albertson and Evens 2000).

San Pablo Bay. Small *populations* of clapper rails are patchy and discontinuously distributed throughout San Pablo Bay in small isolated *tidal marsh* habitat fragments (Collins *et al.* 1994). In 2004 there were between 84 and a few hundred pairs (not individuals) in the San Pablo Bay region (Avocet Research Associates 2004). Highest numbers of clapper rails in San Pablo Bay currently occur in South Gallinas and Hamilton Army Airfield marshes, and at the mouth of

Gallinas Creek (Herzog *et al.* 2006). Clapper rails also occasionally occur along the Petaluma River as far north as Schultz Creek, Lower Tubbs Island, Sonoma Creek area, and along most major *tidal sloughs* that empty into the Napa River (Evens 2000a, 2000b; Collins and Evens 1992; U.S. Geological Survey unpubl. data). In 2006, at least four pairs of clapper rails were detected in *tidal marsh* along San Antonio Creek, just to the north of the Marin Audubon Society's *tidal marsh* restoration site near Neils Island (Marin County; J. Evens *in litt.* 2007). This observation was important since clapper rails have been patchily distributed in the upstream portions of the Petaluma River system.

Clapper rails also occur north to Bull Island on the Napa River (Evens and Collins 1992). They are sparse in the linear strip *marsh* between Highway 37 and San Pablo Bay, most likely due to the lack of dendritic *tidal* creeks. The few clapper rails located in this *marsh* are associated with ditches or natural drainages (U.S. Fish and Wildlife Service unpubl. data).

Surveys conducted in the early 1990s (Evens and Collins 1992, Collins *et al.* 1994, California Department of Fish and Game unpubl. data) indicated a temporary decline in San Pablo Bay clapper rail *populations*. Surveys conducted in the late 1990s indicate that the White Slough area continues to support a moderate number of clapper rails (Evens 2000b). In contrast, rail numbers detected in the Sonoma Creek/Napa Slough area have declined since the early 1990s, from estimates of 13 pairs in 1992 (Evens and Stallcup 1994) to 2 birds detected in 2000 (Evens 2000a).

Suisun Marsh Area. Clapper rails are present sporadically and in low numbers at various locations throughout the Suisun Marsh area (Carquinez Strait to Browns Island, including *tidal* marshes adjacent to Suisun, Honker, and Grizzly bays). Areas where rails have been found recurrently since 1978 include the shoreline marshes from Martinez east to Concord Naval Station, marshes near the mouth of Goodyear Slough (Bahia), Suisun and Hill Sloughs, and the western reaches of Cutoff Slough (Harvey 1980). Surveys in the late 1990s to 2000 indicated that clapper rails were present in marshes associated with Pacheco Creek and Point Edith in Contra Costa County (U.S. Fish and Wildlife Service unpubl. data). Surveys in 2005 found no clapper rails in Suisun Marsh or Point Edith (Herzog *et al.* 2005) and, in 2006, only two clapper rails each at Rush Ranch (Suisun Marsh) and Point Edith (Herzog *et al.* 2006). In addition, this survey identified only two clapper rails at Benicia State Recreation Area (Solano County). Similar sporadic results were found during a multi-year survey by California Department of Fish and Game, in which they detected: no California clapper rails in 2002, eight in 2003, one in 2004, none in 2005, five in 2006, and none in 2007 (California Department of Fish and Game 2008).

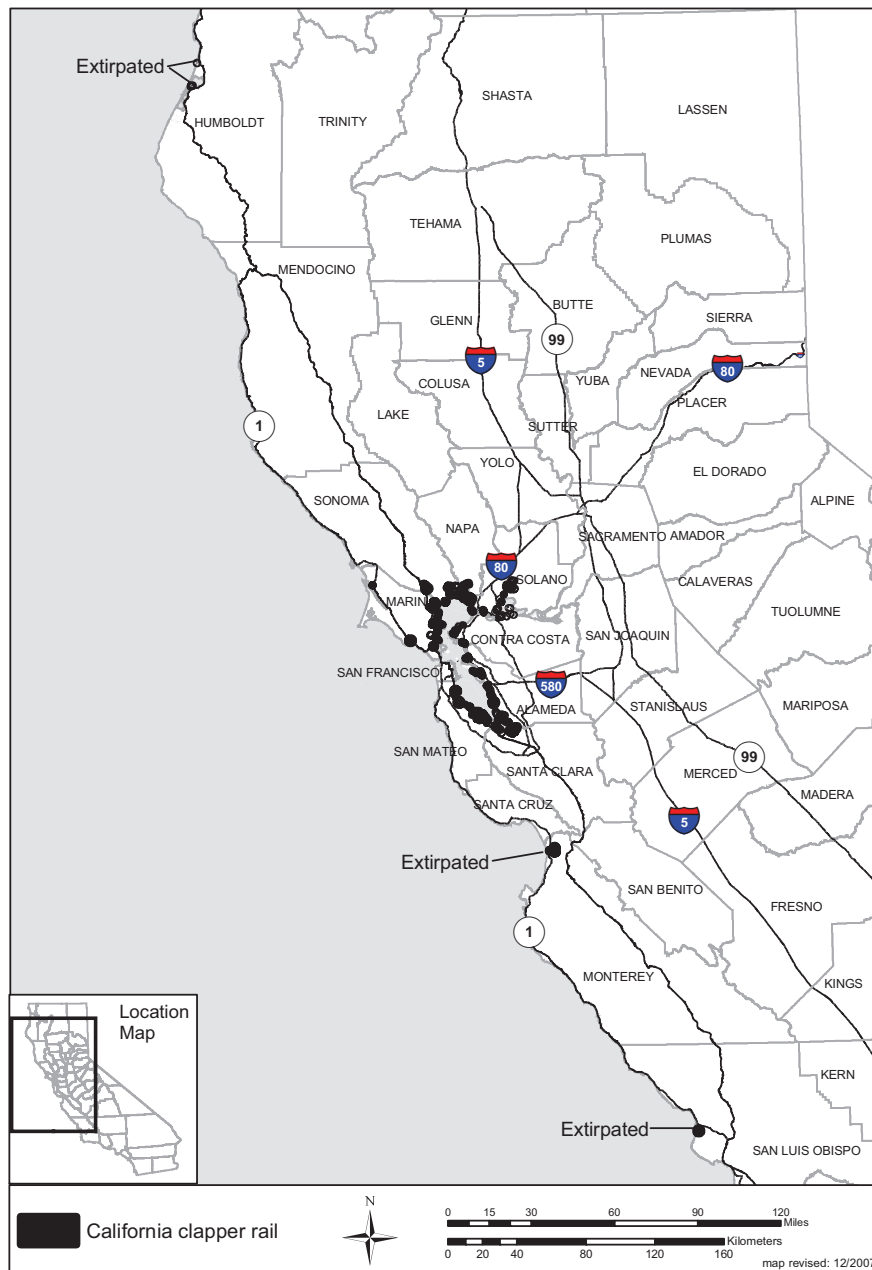


Figure II-9.

Figure II-9. Distribution of California clapper rails, overview

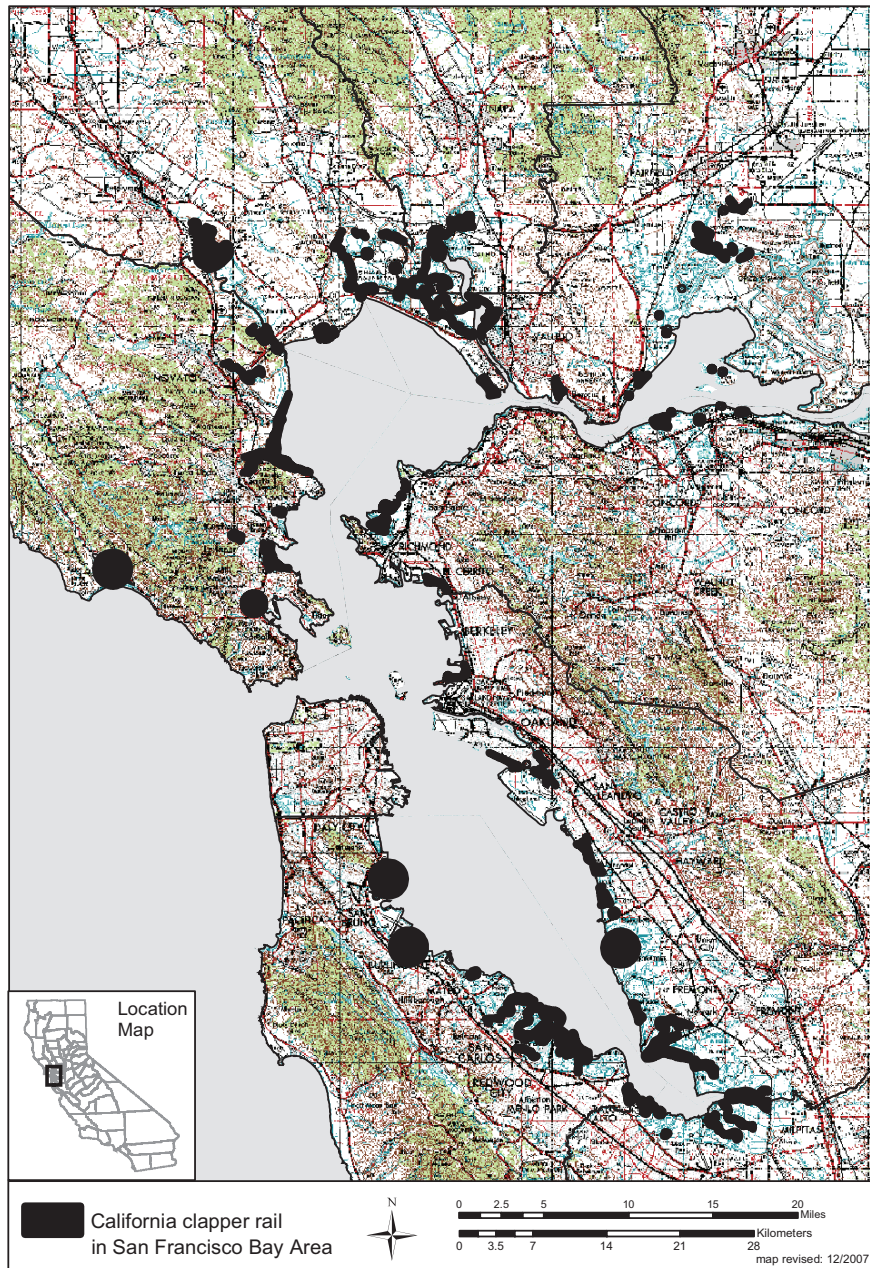


Figure II-10.

Figure II-10. Distribution of California clapper rails, San Francisco Bay

Coastal Areas outside San Francisco Bay. Records of California clapper rails beyond San Francisco Bay are sparse, making *population* status in these areas difficult to track. Few records of clapper rails exist for Humboldt Bay; the last record is from 1947 (Wilbur and Tomlinson 1976). It is unknown whether clapper rails ever bred in Humboldt Bay, and clapper rails observed in that area are widely considered vagrants. Clapper rails had been presumed *extirpated* from Tomales Bay as of 1973, until sightings of single birds were reported there in 1998-2000 (J. Evens *in litt.* 2007). It is unknown whether clapper rails are currently breeding in Tomales Bay, but suitable habitat now exists.

No records of clapper rails have been reported for Morro Bay, San Luis Obispo County, in over 20 years. Clapper rails have not been reported in Elkhorn Slough, Monterey County, since 1980 (Roberson 1993). These three *populations* (Humboldt Bay, Morro Bay, and Elkhorn Slough) are now considered *extirpated*, leaving San Francisco Bay as the last stronghold and breeding *population* of this subspecies.

4) Life History and Ecology

Behavior. In general, clapper rails are secretive and difficult to observe in dense vegetation, but once flushed can frequently be approached (U.S. Fish and Wildlife Service 1984). The U.S. Fish and Wildlife Service considers the California clapper rail sensitive to disturbance, and seeks to minimize human intrusion to occupied marshes, particularly during the breeding season.

When evading discovery, rails typically freeze, hide in small *sloughs* or under overhangs, or run rapidly through vegetation or along slough bottoms (U.S. Fish and Wildlife Service 1984). Rails prefer to walk or run over other forms of locomotion (Ripley 1977, Todd 1986). When flushed, they normally fly only a short distance before landing (Zucca 1954). Clapper rails swim well, although swimming is only used to cross *sloughs* or escape immediate threats at high *tide* (Sibley 1955, Todd 1986).

Clapper rails are diurnally active for 75 to more than 90 percent of the day. Activity peaks in the early morning and late evening (Zembal and Massey 1983, Zembal *et al.* 1989), when rails forage in *marsh* vegetation in and along creeks and *mudflat* edges. Rails often roost at high *tide* during the day (Zembal *et al.* 1989). During the non-breeding season, much of the day is spent roosting and preening.

Courtship. Clapper rails are at least seasonally monogamous, and defend overlapping year-round territories (Zembal *et al.* 1989, Albertson 1995, Garcia 1995). While both sexes advertise for mates, courtship is initiated by the male and involves the male approaching the female with an uplifted tail, pointing his bill to the ground, and swinging it from side to side (Meanley 1985, Albertson and Evens 2000). It is not known whether rails retain their mates between years. Extra-pair copulation is likely, since mated males actively seek unmated advertising females (Zembal and Massey 1985). Males perform most of nest building, and symbolic nest building, wherein males build a nest which is not to be used for actual nesting purposes, may also occur (Meanley 1985). Egg-laying often begins prior to completion of the nest (Eddleman and Conway 1998).

Nesting Phenology. The breeding period of the California clapper rail is prolonged. Pair bonding and nest building are generally initiated by mid-February. Nesting may begin as early as late February or early March (Evens and Page 1983), and extend through July in the South Bay, and into August in the North Bay (DeGroot 1927, U.S. Fish and Wildlife Service unpubl. data). There appears to be a break in nesting between mid-May through late June in the North Bay, a period that corresponds to the highest summer *tides* (Evens and Page 1983). Two peaks in nesting activity occur, a greater peak between mid-April and early-May and a lesser peak between late-June and early-July (DeGroot 1927, Applegarth 1938, Gill 1972, Harvey 1988). The second nesting peak has been interpreted as attempts by late nesters (DeGroot 1927), second attempts after initial nesting failures (Gill 1972), or second broods (Wilbur and Tomlinson 1976).

Rails frequently build several nest platforms, but use only one for incubation (Applegarth 1938, Gill 1972, Wilbur and Tomlinson 1976). Both sexes share in incubation, which lasts from 18-29 days (Taylor 1996). Eggs are approximately 45 millimeters (1.77 inch) in length, and light tan or buff-colored with cinnamon-brown or dark lavender spotting concentrated at the broader end. Estimates of California clapper rail clutch size range from 5-14 eggs (DeGroot 1927, Gill 1972). Mean clutch sizes of 7.1 (U.S. Fish and Wildlife Service unpubl. data) to 7.5 (Foerster *et al.* 1990) have been reported. Hatching is generally *synchronous*, but occasionally eggs hatch one to several days apart (R. Zembal pers. comm., U.S. Fish and Wildlife Service unpubl. data). Defense of the nest site intensifies as hatching approaches (Applegarth 1938, U.S. Fish and Wildlife Service unpubl. data). Hatching requires approximately 48 hours to complete after breaking through of the shell (Johnston 1956a). Chicks soon depart the incubation nest, and one to three brood nests are typically constructed nearby (Applegarth 1938, Johnson 1973). Brood nests are high *tide* refuges for young rails, and consist of a platform of woven stems without a substantial canopy (Harvey 1980). These may also be used as gathering points and resting places for the young. Adults remain with the chicks to forage with them for up to 5 to 6 weeks (Applegarth 1938, Meanley 1985).

Nest Site. Rails require an intricate network of *sloughs* to provide abundant invertebrate *populations* (Grinnell *et al.* 1918, DeGroot 1927, Harvey 1988, Collins *et al.* 1994) and escape routes from predators, particularly for vulnerable flightless young (Taylor 1894, Adams 1900, DeGroot 1927, Evens and Page 1983, Foerster *et al.* 1990, Evens and Collins 1992). In addition, the small natural *berms* along *tidal* channels with relatively tall vegetation, such as *Grindelia stricta* (gumplant), provide elevated nesting substrate.

Nests must be built at an elevation that protects the bowl from complete inundation during high *tides* (Evens and Collins 1992, Collins *et al.* 1994). However, some nests are built directly on the ground. If a nest settles or gets wet, the adults may add additional materials such that a minimum elevation above the *tides* is maintained. Inundated nests result in abandonment and failure (U.S. Fish and Wildlife Service unpubl. data). Zucca (1954) proposed that late nesting attempts resulted from interruption of earlier attempts by high *tides*.

California clapper rails are relatively indiscriminate in their choice of nesting substrate, and prefer to use the tallest cover regardless of plant species (Garcia 1995). However, rails typically nest in the upper-middle *tidal marsh* plain or high *tidal marsh* zones, not upland habitat

transition zones bordering *tidal marsh*. Vegetation must be 50 centimeters (19.7 inches) high or greater near mean high water to allow for nest concealment and prevent *tidal* inundation. Robust *Sarcocornia pacifica* (pickleweed) or *Grindelia* vegetation is usually selected for nest locations in San Francisco Bay. Shorter vegetation may be used at higher *marsh* elevations (Albertson and Evens 2000). Plant species used for nest construction includes *Spartina* spp. (cordgrass), *Sarcocornia*, *Grindelia*, *Distichlis spicata* (saltgrass), *Scirpus* spp. (bulrushes), *Typha* (cattails), *Spartina wrack*, *Jaumea carnosa* (fleshy jaumea), lodged tumbleweeds, and other drift materials (DeGroot 1927, Zucca 1954, Gill 1972, Harvey 1980, Foerster *et al.* 1990, Garcia 1995).

Clapper rail nests consist of a platform surrounded by vegetation that has been pulled together to form a canopy. In the South Bay, most nests are located in *Grindelia* and *Sarcocornia*, with platforms constructed from *Spartina* and *Sarcocornia* (Harvey 1980, Foerster *et al.* 1990, U.S. Fish and Wildlife Service unpubl. data). Foerster *et al.* (1990) found evidence of preferential use of *Spartina* in nest platforms. In the *brackish* reaches of the northern San Francisco Bay Estuary, many clapper rail nests are located in *Scirpus*. North Bay platforms typically consist of *Sarcocornia*, mixed *Distichlis* and *Sarcocornia*, or *Scirpus* (Garcia 1995, Albertson and Evens 2000, U.S. Fish and Wildlife Service unpubl. data). Throughout the bay, variations in nest materials used by clapper rails have been reported (DeGroot 1927, Zucca 1954, Gill 1972, Harvey 1980, Foerster *et al.* 1990, Garcia 1995).

Productivity. Reproductive success of the California clapper rail is much reduced below the natural potential (Schwarzbach *et al.* 2006). Information on reproductive success (hatch, nest, and fledge success) is available from three studies conducted in the South Bay (**Table II-2**), and one study in the Central Bay. In a 1980 investigation of Dumbarton, Ideal, and Mowry Marshes, Harvey (1988) found hatching success was 38 percent. However, in a follow-up study, Foerster *et al.* (1990) reported 19 percent hatching success at Dumbarton and Mowry. In both investigations predation was reported to account for only a third of the lost eggs. Additional investigations in Faber Marsh in 1991, and in Faber, Mowry, Laumeister, and Greco Marshes in 1992, found a hatching success of 43 percent (U.S. Fish and Wildlife Service unpubl. data). Predation accounted for a loss of 38 percent of the eggs, flooding for 1.4 percent, abandonment for 3.3 percent, 16 percent of the eggs were non-viable, and the fate of 1.3 percent of the eggs was unknown. South Bay marshes evaluated in 1991-1992 produced 2.5 hatched eggs per nesting attempt (**Table II-3**). In a 1998-1999 study in Central Bay marshes, clapper rail hatching success was 51.6 percent in Wildcat Marsh and 26.9 percent at Muzzi Marsh. Central Bay marshes evaluated in 1998-1999 produced 1.89 young per nesting attempt.

Table II-2. Summary of California clapper rail reproductive success. South Bay data are from 1980, 1988, and 1991- 1992; North Bay data are from 1998-99.				
	1980 ^a	1988 ^b	1991-92 ^c	1998-99 ^d
Total number of nests found	50	29	na	Na ^e
Number of active nests monitored	26	24	71	18
Mean clutch size	7.3	7.5	7.0	6.7
Total number of eggs	189	155	431	98
Eggs hatched	71	29	177	34
Eggs lost to predators	63	51	164	38

Eggs unhatched ^f	34	36	71	25
Eggs disappeared	21	39	6	1
Nest success (percent in parentheses)	28 (56)	6 (32)	33 (47)	7 (39)
Nest failure (percent in parentheses)	16 (32)	11 (46)	38 (53)	11 (61)
Nest fate unknown	6	7	26	3
^a Harvey 1980, study sites = Dumbarton (n=27), Mowry (n=18), Ideal (n=5) ^b Foerster <i>et al.</i> 1990, study sites = Dumbarton and Mowry ^c U.S. Fish and Wildlife Service unpubl. data, 1991 study site = Faber (n=16); 1992 study sites = Faber (n=4), Greco (n=20), Mowry (n=10), and Laumeister (n=20) ^d U.S. Fish and Wildlife Service unpubl. data, study sites = Corte Madera (n=11), Wildcat (n=7), Petaluma (n=1) ^e Not available ^f Includes eggs lost to flooding and fail-to-hatch eggs				

Table II-3. Clapper rail nest fate summary table. South Bay data are from 1992; North Bay data are from 1998-1999 (U.S. Fish and Wildlife Service unpubl. data).

	Nests	Total Young	Mean Clutch Size	Young/ Nest Attempt	% Hatchability ^a	% Hatch Success ^b	% Nest Predation (#)	% Nest Success (#)	% Nest Flooding (#)
South Bay Total	71	177	6.95	2.5	71.3	42.9	47.9	46.5	1.4
Faber	14	41	6.75	2.9	71.9	46.6	21.4 (3)	50.0 (7)	7.1 (1)
Greco	20	61	6.75	3.1	75.6	45.2	50.0 (10)	60.0 (12)	0 (0)
Laumeister	26	33	6.73	1.3	62.5	25.1	61.5 (16)	30.8 (8)	0 (0)
Mowry	11	42	7.60	3.8	75.0	54.5	45.5 (5)	54.5 (6)	0 (0)
North Bay Total	18	34	6.66	1.9	65.0	34.7	41.5	42.2	5.5
Heerdt	11	18	6.90	1.6	69.0	26.9	54.5 (6)	27.3 (3)	9 (1)
Wildcat	7	16	6.25	2.2	60.0	51.6	28.6 (2)	57.1 (4)	0 (0)
Overall Total	89	211	6.89	2.4	70.6	38.2	47.2	44.9	2.3
^a Hatchability is calculated as the number of eggs hatched / the number of eggs incubated to term (<i>i.e.</i> available to hatch). ^b Hatch success is calculated as the number of eggs hatched per nest / clutch size.									

Hatchability for clapper rails in San Francisco Bay varies with *marsh*. In the 1991-1992 South Bay investigations, hatchability ranged from 62.5 to 75.6 percent, with Laumeister having the lowest hatchability. Hatchability at Central Bay marshes in 1998-99 was 60 percent and 69 percent for Wildcat and Heerdt, respectively.

Normal hatch success and hatchability of clapper rail eggs is much higher (Zembal and Massey unpubl. data, Jorgensen 1975). A study of clapper rails in New Jersey indicated an 87.3 percent hatch success (Kozicky and Schmidt 1949). The hatching success and hatchability of the California clapper rail is clearly impaired. Reasons for low hatchability of eggs could include contamination, loss of *genetic* diversity, and reduced incubation of eggs due to disturbance. There is reason to believe that contamination may be the cause of some of the observed impairment in hatchability (**Appendix E**).

Currently, no data are available on fledge success for California clapper rails.

Feeding Ecology. The clapper rail is an omnivore with a relatively broad feeding niche. Animal matter has been consistently emphasized as a major component of the diet (Moffitt 1941, Heard 1982, Zembal and Fancher 1988). Food items found in California clapper rails stomachs include introduced ribbed horse mussel (*Ischadium demissum*), spiders (*Lycosidae* spp.), clams (*Macoma balthica*), yellow shore crabs (*Hemigrapsus oregonensis*), amphipods (shrimp-like crustaceans), *Nereis vexillosa*, polychaetes (a class of annelid worms), and striped shore crab (*Pachygrapsus crassipes*; Williams 1929, Applegarth 1938, Test and Test 1942, Varoujean 1972). Rails occasionally have been seen capturing and consuming rodents, particularly during higher tides; small birds are also occasionally taken (Spendelow and Spendelow 1980, Jorgenson and Ferguson 1982).

Territoriality/Site Fidelity. Clapper rails exhibit strong territorial defense, particularly during the late winter and early breeding seasons (Williams 1929, Albertson 1995, Garcia 1995). Territoriality weakens during extreme high tides when cover is limited, and during the post-breeding season. Rails have been observed in groups of 10 or more during winter high tide surveys (U.S. Fish and Wildlife Service unpubl. data). Little information is available on interspecies aggression in rails, though a California clapper rail has been observed successfully fending off a northern harrier (*Circus cyaneus*) from a brood and, on another occasion, itself at the Corte Madera Creek mouth, Marin County (J. Evens *in litt.* 2009).

Clapper rails generally exhibit strong site fidelity (Albertson 1995) although they do disperse. A banding study in the mid-1980s revealed the limited movement of rails in the South Bay, with 78 percent of resightings within 500 m (1,641 ft) of the original capture site (U.S. Fish and Wildlife Service unpubl. data).

Home Range. A 1991-1992 radiotelemetry study in south San Francisco Bay indicated an average home range of 4.7 hectares (11.6 acres) and an average core use area of 0.9 hectare (2.2 acres; Albertson 1995). Home ranges were maintained throughout the year, but varied among marshes and seasons. During the breeding season, average home ranges expanded from 2.9 hectares (7.1 acres) in January-February, to 3.7 hectares (9.1 acres) in May-July.

Home range size and site fidelity may be impacted by disturbance. Albertson (1995) documented a rail abandoning its territory shortly after a repair crew worked on a nearby transmission tower. The bird did not establish a stable territory within the duration of the breeding season, but eventually moved closer to its original home range several months after the disturbance. In contrast, clapper rails have been documented nesting near trails and dikes used by pedestrians/vehicles in Elsie Romer and Cogswell Marshes (J. Didonato pers. comm.). The reproductive success of these clapper rails is unknown.

Garcia (1995) evaluated the use of call count surveys for determining clapper rail territory size in Marin County, and found that territory size is underestimated using this approach. This is because rails call from core areas that are less than 35 percent of the total territory area used during the breeding season (Eddleman 1989, Conway *et al.* 1993). However, multiple call count

surveys conducted between mid-January and mid-April significantly increase the accuracy of *population* estimates of clapper rails compared to single call count surveys (Garcia 1995).

Density. Density estimates are typically reported as the number of rails over the total acreage of the *tidal marsh* parcel. Because this method does not discount areas that are not suitable habitat, density estimates for clapper rails may underestimate the density of rails in appropriate habitat..

Numerous studies (Applegarth 1938, Gill 1979, Harvey 1988, Foerster *et al.* 1990, Collins *et al.* 1994) provide data on rail breeding densities in the South Bay (**Table II-4**). Estimates of clapper rail wintering (non-breeding) densities are variable and limited (Gill 1979, Moss 1980; Harvey 1980, 1981; Foerster 1989).

Dispersal. Post-breeding dispersal has been documented during the fall and early winter (Lindsdale 1936, Orr 1939, U.S. Fish and Wildlife Service unpubl. data, Albertson 1995). There is no clear evidence of migratory behavior in the California clapper rail. However, infrequent long distance dispersal does occur. Vagrant rails have been found in areas not known to support individuals throughout the year, such as the Farallon Islands (Bryant 1888), the rocky shores of Pacific Grove (Kimball 1922), and Pescadero Marsh (Orr 1942). These birds have been found primarily in late summer and fall, and are assumed to be dispersing subadults.

Survivorship. The only estimates of annual adult California clapper rail survivorship were relatively low, ranging from 0.49 to 0.52 (Albertson 1995). These are similar to survival estimates reported for the Yuma subspecies (Eddleman 1989). Increased predation occurs during extreme winter high *tides*, probably due to increased movement of rails at this time when little cover is available (Albertson and Evens 2000). Adult survivorship has been suggested as the key *demographic* variable associated with survival of clapper rail *populations* (Foin *et al.* 1997).

Predators. Predators known to prey on clapper rails and their eggs include the native gopher snake (*Pituophis melanoleucus*), great blue heron (*Ardea herodias*), red-tailed hawk (*Buteo jamaicensis*), peregrine falcon (*Falco peregrinus*), northern harrier (*Circus cyaneus*), barn owl (*Tyto alba*), great horned owl (*Bubo virginianus*), short-eared owl (*Asio flammeus*), common raven (*Corvus corax*), raccoon (*Procyon lotor*), and California ground squirrel (*Spermophilus beechyii*) (Johnston 1956b). *Non-native* predators identified to date include the Norway rat (*Rattus norvegicus*), red fox (*Vulpes vulpes*), feral cat (*Felis catus*) and feral hogs (*Sus scrofa*; B. Grewell *in litt.* 2006b). Adult clapper rails may be preyed upon by all of the above species except gopher snakes, ravens, raccoons, ground squirrels, and rats, which prey on eggs or chicks.

Table II-4. Estimates of California clapper rail breeding densities in San Francisco Bay.				
Site Name	Year	Density (rails/hectare)	Location ^a	Source
Dumbarton	1986	1.47	South Bay	Harvey 1988
	1988	0.64		Foerster <i>et al.</i> 1990
Mowry	1986	0.89	South Bay	Harvey 1988
	1988	0.26		Foerster <i>et al.</i> 1990
Audubon	1988	0.18	South Bay	Foerster <i>et al.</i> 1990
Ideal	1986	0.69	South Bay	Foerster <i>et al.</i> 1990

Central	1993	0.33	Central Bay ^b	Collins <i>et al.</i> 1994
Petaluma River	1993	0.26	North Bay ^c	Collins <i>et al.</i> 1994
Sonoma Creek	1993	0.18	North Bay ^d	Collins <i>et al.</i> 1994
Napa River	1993	0.23	North Bay ^e	Collins <i>et al.</i> 1994
Carquinez Strait	1993	0.03	North Bay ^f	Collins <i>et al.</i> 1994
Suisun Bay	1993	0.09	North Bay ^g	Collins <i>et al.</i> 1994
Grizzly Bay	1993	0.09	North Bay ^h	Collins <i>et al.</i> 1994
^a South Bay density estimates used rope drags; North Bay density estimates used call counts. ^b Central Bay included Richardson Bay, Muzzi, Corte Madera, Creekside, Gallinas, Hamilton, Point Pinole, and Wildcat marshes. ^c Petaluma River included sites at the river mouth, Novato Creek, Black John Slough, Mira Monte Slough, Tule Slough, and Shultz Slough. ^d Sonoma Creek included sites at the creek mouth, Second Napa Slough, Hudeman Slough, and Wingo. ^e Napa River included sites at White Slough, Wilson Avenue, River Park, Boxer Marsh, Coon Island, Fagan Slough/Bull Island, Napa Town, and Mare Island Point. ^f Carquinez Strait included Southampton/Benicia and Martinez. ^g Suisun Bay included sites at Pacheco Creek, Point Edith, Port Chicago, and Antioch. ^h Grizzly Bay included sites at Bahia, Goodyear Slough, the mouth of Suisun Slough, Cutoff Slough, Mallard Slough, Hill Slough/Union Creek, Navy Point, Boynton Slough, and Peytonia Slough.				

Of these predators, raptors, Norway rats, and red fox are the most significant (DeGroot 1927, Foerster 1989, Albertson 1995, Harding *et al.* 1998). Studies in 1991-1992 found a negative correlation between red fox numbers and rail densities (Harding *et al.* 1998, Albertson 1995). The most severe rail *population* declines and highest fox numbers were found in the East Bay marshes (*e.g.*, Dumbarton, Mowry, Ideal, and Calaveras). Winter airboat surveys in 1992-1993 documented a clapper rail *population* increase in many South Bay marshes in apparent response to predator control that began in 1991 (Harding *et al.* 1998).

The temporary decline in San Pablo Bay clapper rail *populations* in the early 1990s (Evens and Collins 1992, Collins *et al.* 1994, California Department of Fish and Game unpubl. data) may have occurred in response to invasion by red fox, wet winters that caused extreme flooding of *tidal* marshes and encouraged the growth of *Scirpus maritimus* (alkali-bulrush) to the detriment of *Spartina foliosa* (Pacific cordgrass) habitat in the low *marsh*, or a combination of factors. The additional predation pressure from red fox invasion with a resulting increase in failed nests may have increased the importance of the second, mid-summer peak in nesting activity.

5) *Habitat Characteristics/Ecosystem*

Throughout their distribution, California clapper rails occur within a range of salt and *brackish* marshes (Harvey *et al.* 1977). In south and central San Francisco Bay, and along the perimeter of San Pablo Bay, rails typically inhabit salt marshes dominated by *Sarcocornia pacifica* and *Spartina foliosa*. *Spartina* dominates the middle *marsh* zone (*marsh* plain) throughout the south and Central Bay (DeGroot 1927, Hinde 1954, Harvey 1988). *Sarcocornia* dominates the upper *marsh* zone throughout the South and Central Bay, with *Distichlis spicata*, *Jaumea carnosa*, and *Frankenia salina* (alkali-heath). *Grindelia stricta* var. *angustifolia* occurs along the upper edge of *tidal sloughs* throughout the entire San Francisco Bay Estuary. The marshes of Humboldt

Bay, Morro Bay, and Elkhorn Slough historically have not supported *Spartina*. Vegetation at these locations has been dominated by *Sarcocornia pacifica* and *Distichlis spicata*.

In the North Bay, clapper rails also occur in *tidal brackish* marshes that vary significantly in vegetation structure and composition, ranging from salt-*brackish marsh* to fresh-*brackish marsh* transitions. *Scirpus maritimus* (alkali bulrush), an indicator of salt-*brackish marsh* transitions, is sub-dominant to dominant in low *marsh* and lower middle *marsh* plains. *Scirpus acutus* and *Scirpus californicus* (tules), *Scirpus americanus* (Olney's bulrush), and *Typha* spp. dominate the low *marsh* zone of fresh-*brackish marsh* transitions, while fresh-*brackish marsh* plain vegetation is a diverse, patchy mixture of dominant *Distichlis*, *Jaumea*, salt rush (*Juncus balticus*, *Juncus lesueurii*), and numerous native and *non-native* herbs, grasses, and sedges. *Grindelia stricta* var. *angustifolia* (and its hybrid *Grindelia x paludosum* in Suisun Marsh) is the widespread dominant of high *marsh* vegetation in *brackish* marshes today, but it occurs with other tall, dense sub-shrubby or herbaceous native vegetation along *marsh* edges and creek banks, such as *Baccharis douglasii* (salt *marsh* baccharis), *Euthamia occidentalis* (goldenrod), *Achillea millefolium* (yarrow), *Scrophularia californica* (bee-plant), and asters (*Aster lentus*, *Aster chilensis*, and intermediates, *Aster subulatus* var. *ligulatus*; now uncommon). The historically diverse high *brackish marsh* vegetation probably provided ample high *tide* flooding refuges for clapper rails.

Use of *brackish* marshes by clapper rails is largely restricted to major *sloughs* and rivers of San Pablo Bay and western Suisun Marsh, and along portions of Coyote Creek in south San Francisco Bay. In *brackish* marshes, other rail species such as Virginia rail and sora rail (*Porzana carolina*) are typically more common than clapper rails. The frequency of clapper rail sightings in Suisun Marsh was noted to increase during drought periods, which are associated with elevated *marsh salinity* (Harvey 1977). Clapper rails were not reported from Suisun Marsh in the 19th and early 20th centuries. However, they have persisted in Suisun Marsh even after above-average rainfall and very low channel *salinity* in the 1990s, when *tidal* marshes there developed a fresh-*brackish* vegetation (Estrella *in litt.* 2007).

Clapper rails have rarely been recorded in nontidal *marsh* areas. Small numbers have been detected calling during the breeding season in a *diked Sarcocornia* habitat in Crittenden Marsh, Santa Clara County (Orton-Palmer and Takekawa 1992) and in Richardson Bay, Marin County (J. Evens *in litt.* 2009).

Rail foraging and refugial habitat encompasses the lower, middle, and high *marsh* zones, as well as the adjacent transitional zone. Lower and middle *marsh* zones provide foraging habitat at low *tide*. Small *tidal* channels (*i.e.*, first- and second-order) with dense vegetation covering the banks are particularly important habitat features (Keldsen 1997, Garcia 1995). These provide important foraging habitat and hidden routes for travel in close proximity to nesting habitat. Higher *marsh* areas (high *marsh* and transitional zones) with dense vegetation are used for nesting and high-*tide refugia* (DeGroot 1927, Harvey 1988, Foerster *et al.* 1990, Evens and Collins 1992, Collins *et al.* 1994). Within *tidal* marshland in portions of north San Francisco Bay, the abundance of California clapper rails is positively correlated with channel density or the total length of channel per unit area of marshland (Garcia 1995, Evens and Collins 1992, Collins *et al.* 1994, Foin *et al.* 1997). Keldsen (1997) found that rails prefer locations with a greater number of *tidal* creeks, *Grindelia* shrubs, and higher elevations.

The quality of a *marsh* strongly influences the density of rail *population* it can support (Albertson 1995, Garcia 1995). Physical habitat characteristics critical to clapper rails include *marsh* size, location relative to other marshes, presence of buffers or transitional zones between marshes and upland areas, *marsh* elevation, and *hydrology* (Collins *et al.* 1994, Albertson 1995). Denser rail *populations* exist where the habitat patch size is greater than 100 hectares (247 acres; Collins *et al.* 1994). Currently, there are fewer than 15 such patches in the San Francisco Bay Estuary (Albertson and Evens 2000).

6) Critical Habitat

No critical habitat has been designated for the California clapper rail.

7) Reasons for Decline and Threats to Survival

Most species covered in this draft recovery plan are threatened by similar factors because they occupy the same *tidal marsh* ecosystem. These general threats, faced by all covered species, are discussed in greater detail in the Introduction section of this draft recovery plan (section I.D.). Specific threats to California clapper rail are described below.

Conversion of *tidal marsh* on a large scale began in the late 1800s. In the South Bay, *tidal marsh* was *diked* and drained primarily for urban and industrial development. In the North Bay (San Pablo and Suisun included), reclaimed land was used for grazing, agriculture, and duck clubs (Goals Project 1999). Coupled with the unprecedented habitat loss of the mid-1800s was equally unprecedented hunting pressure. Kennerly (1859) indicated that clapper rails were one of the most numerous birds sold in San Francisco markets during the mid-1800s. Up to 1889, bagging up to 200 clapper rails per hunting trip was not uncommon (Grinnell *et al.* 1918). By 1894, clapper rail *populations* had noticeably declined (Taylor 1894), and some people in the South Bay were advocating a temporary closure of the rail hunting season (Cohen 1899). By 1902, clapper rail numbers had dropped precipitously due to simultaneous habitat loss and hunting pressure. The annual closed summer season remained in effect, but was insufficient protection. *Tidal marsh* conversion concentrated the birds in smaller areas, greatly facilitating fall hunting (Grinnell *et al.* 1918).

The Federal Migratory Bird Law, passed in 1913, was designed to stop illegal shipment of migratory birds across state lines. The Federal Migratory Bird Act, passed in 1918, decreed that all migratory birds and their parts were fully protected. Neither legislation, however, prevented the conversion of *tidal marsh*, which continued in the 1900s. The red fox was introduced to San Francisco Bay in 1980 and resulted in another precipitous clapper rail *population* crash in the mid-1980s (Harding *et al.* 1998).

Habitat Loss. The loss of coastal wetland habitat to urban and industrial development has been extensive in California, with a 90 percent total loss of all wetlands since settlement of the region (Goals Project 1999). Habitat loss has dramatically slowed since the rail was listed in 1970, but ongoing disturbance and degradation precludes or reduces occupation of much of the remaining potential habitat by California clapper rails. Remaining habitat has been fragmented by *levee*

systems that reduce and isolate patches of habitat, reduce/eliminate high *marsh* and refugial habitat, and make habitat accessible to predators and human disturbance. Habitat has been filled, subjected to many contaminants, converted to less suitable vegetation conditions by fresh wastewater discharges, and submerged by land *subsidence* caused by agricultural practices and groundwater overexploitation. Spread of *non-native* saltmarsh vegetation threatens to degrade remaining habitat and reduce the benefits of future habitat restoration.

Loss of upper *marsh* vegetation has greatly reduced available habitat throughout the range of the subspecies. Most marshes in south San Francisco Bay are adjacent to steep earthen *levees* that have all but eliminated upper *marsh* vegetation and reduced available cover for rails during winter flood *tides*. In Suisun Marsh, high *marsh* vegetation has been eliminated by diking and livestock grazing.

In addition to the problems associated with landscape alteration caused by development, California coastal wetlands are expected to be subject to the effects of global sea level rise and climate change due to global warming (see Introduction section I.D.e). The effects of past *subsidence* of *marsh* plain relative to mean *tidal* level, particularly in the South Bay (Atwater *et al.* 1979), are likely to be amplified by rising *tidal* levels.

Habitat Degradation. Other than outright habitat loss due to *marsh* reclamation, significant historic degradation to clapper rail habitat quality in remaining *tidal* marshes is caused by numerous human-caused physical and biological changes in the San Francisco Bay Estuary *tidal* marshes, including:

- (1) Construction and maintenance of *dikes* in *tidal* wetlands—many adverse effects stem from these actions, including
 - a. *marsh* fragmentation and reduction to small isolated marshes
 - b. reduction in quality, distribution, and abundance of critical sub-habitats, such as high *tide refugia*
 - c. reduction and simplification of natural *tidal* creek and *levee* networks by *dikes* and flood control channels
 - d. locally excessive sedimentation induced by diking of *tidal* creeks
 - e. establishment of extensive non-*tidal* predator *corridors*, perches, and nest/den sites
 - f. *marsh subsidence* and submergence due to groundwater overexploitation
- (2) Replacement of *tidal refugia* along landward *marsh* edges with unbuffered urban edges.
- (3) Conversion of salt *marsh* to *brackish-fresh marsh* by urban fresh wastewater discharges.
- (4) Structural habitat change caused by *non-native* plant invasions (particularly *Spartina alterniflora* in low *marsh* and *Lepidium latifolium* in high *marsh*).
- (5) Increased predation by attracted avian and mammalian predators due to availability of man-made structures. Electrical towers and nearby buildings may be used for nesting and roosting of avian predators. Boardwalks may be used for roosting, in addition to serving as routes of access into the *marsh* interior for mammalian predators.

- (6) Increased disturbance from recreational access, including humans and dogs.
- (7) Reduced habitat quality and increased predation pressure from litter and debris.
- (8) Contamination of *marsh sediments*, which may impact clapper rails directly or indirectly. Potential direct effects include toxicity to adults, chicks, or embryos. Potential indirect effects include reduced prey quality, quantity, and availability, and altered vegetation structure/composition for nesting and sheltering (see **Appendix E**).

Few of these causes of habitat degradation are independent of one another; they interact and mutually amplify. For example, construction and subsequent maintenance of a *dike* restricts *tidal* circulation, concentrates impacts of any fresh wastewater discharges, provides predator *corridors* and nest/den sites, compresses *tidal* refugial vegetation to a narrow strip, and promotes ruderal (weedy) vegetation. It may also mobilize contaminants buried in *marsh sediments*. Further, the presence of a *dike* may provide recreational access for people and their pets, which results in increased disturbance and potential litter problems. Rodents attracted to the litter, and provided access and nest sites by *dikes*, will result in added predation pressure on clapper rails.

Dikes and other *tidal* barriers have led to widespread degradation and loss of clapper rail habitat in the San Francisco *estuary* and elsewhere. *Dikes* typically are maintained by clamshell dredges that deposit bay spoil material on the tops and sides of the *dikes*, periodically reducing or eliminating high *tide* escape cover and nesting habitat. *Dike* systems isolate salt marshes in areas too small to develop the complex *tidal* drainage networks needed for productive clapper rail habitats. Natural *tidal* marshes are the result of a delicate balance between *sediment* erosion and deposition. A shift in the local *sediment* budget, whether from natural or artificial factors, will impact clapper rail' habitat.

Many of the *tidal* marshes in the bay are relatively small fragments, and the presence of *dikes* facilitates predator access across the entire site. This is particularly true for the linear/strip marshes prevalent in the South Bay. *Dikes* allow predators to travel miles out into baylands that would otherwise be naturally isolated from terrestrial predators. Mammalian predators, especially red foxes and rats, use *levees* as movement *corridors* and denning/nesting sites, as described below under *Predation*. Any clapper rail nests located close to *dikes* are therefore subject to higher predation pressures. The red fox is a highly efficient predator of rail eggs, chicks, and adults in the South Bay. Red foxes do not typically travel far from the *dikes*, however, so large marshes generally have lower rates of fox predation than small marshes.

Fragmentation. Other threats result indirectly from *dikes* due to the breakdown of *tidal marsh* habitat into relatively small, discontinuous, narrow fragments. Generally, extinction rates increase as habitat size decreases and distance from neighboring *populations* increases (MacArthur and Wilson 1967). As remaining habitat units decrease in size, *edge effects* become increasingly important. Smaller units have less space available to buffer adverse impacts from outside influences, such as predation, human disturbance, or chemical contamination. In addition, *catastrophic* mortality from chance environmental events, such as flooding, is a severe threat to the long-term survival of small, isolated *populations* (Schonewald-Cox *et al.* 1983).

Isolation of small local *populations* increases chances of *inbreeding*. The breeding of closely related individuals can cause *genetic* problems in small *populations*, particularly the expression of deleterious *genes* (*inbreeding* depression). Individuals and *populations* possessing deleterious *genetic* material are less able to cope with environmental conditions and adapt to environmental change. Furthermore, small *populations* are subject to the effects of *genetic* drift (random loss of *genetic* variability). *Populations* that undergo extreme declines and rebound from a small number of survivors are particularly vulnerable to *inbreeding* depression. Clapper rails in San Francisco Bay suffer from both risks. Loss of *genetic* variability may also limit the ability of individuals and *populations* to successfully respond to environmental stresses. Overall, these *genetic* factors affect *population* fitness and the likelihood of survival of the species.

Buffer zones: Prior to settlement of the bay area by Europeans, *tidal* baylands graded into a transitional zone of low-lying moist grassland and *vernal pool* habitat, and then into upland areas (Goals Project 1999). Much of the historical development around the bay did not allow for buffer zones between urban/industrial areas and *tidal* marshes. Buffer zones reduce the impact of adjacent incompatible land use. The most appropriate type of buffer for the protection of clapper rails is an area of terrestrial/*marsh ecotone*. These areas provide two primary benefits to adjoining wetlands by (1) absorbing and deflecting disturbances originating in upland areas, and (2) providing upland *refugia* during high *tide* and flood events, both of which ultimately influence habitat quality and carrying capacity of *tidal* marshes for clapper rails. Appropriately sized and structured buffer zones are a critical component of clapper rail habitats in urbanized settings.

Wastewater discharges: Wastewater discharges that alter natural *salinity* levels in *tidal* waters can adversely affect clapper rail *populations* and other species. Since about 1970, *freshwater* discharges on the order of 120 million gallons/day from the San Jose Water Pollution Treatment Plant, have led to the conversion of approximately 120 hectares (300 acres) of former salt *marsh* to fresh and *brackish marsh* at the southern end of San Francisco Bay along Coyote Creek and adjoining *sloughs* of the Santa Clara Valley (H.T. Harvey and Associates 1997). *Marsh* conversion may lower the habitat quality and carrying capacity of *tidal* marshes to support clapper rails, as evidenced by lower *population* and nesting densities recorded in *brackish* marshes than salt marshes (H.T. Harvey and Associates 1989).

Non-native vegetation: Some introduced plants, particularly *Lepidium latifolium* (perennial pepperweed) and *Spartina alterniflora*, appear to pose threats to habitat quality for clapper rails. The rapid spread of *Lepidium latifolium* throughout thousands of acres of *brackish* marshes and *brackish* high *marsh* edges in Suisun, San Pablo, and south San Francisco bays may interfere with seedling establishment of *Grindelia*, a tall native evergreen sub-shrub used by clapper rails for high *tide* cover and nesting substrate in high *marsh*. *Lepidium latifolium* establishes poor above-ground winter cover as it is leafless and provides little cover during high winter *tides*. Spreading rhizomatously and by seed, it may displace *Sarcocornia pacifica* and other plants in some locations. The extent to which this species may affect clapper rails and other native species has not been investigated. Further study is needed to assess the effects of *Lepidium latifolium* in the marshes of San Francisco Bay.

Spartina alterniflora colonizes *mudflats*, *pans*, and *slough* channels. Once established it causes decreased water flow, increased *sediment* deposition, and infill. The net result is an increase in elevation of the area, exclusion of native *Spartina foliosa*, and replacement of diverse native vegetation with a monotypic stand that lacks vertical complexity. Small *tidal* creeks, *mudflats*, and *pans* are at significant risk of infilling with *Spartina alterniflora*. This could result in loss of foraging habitat and movement *corridors* for clapper rails. It is important to note that the habitat requirements of clapper rails include both dense cover for nesting and access to low *marsh* (*tidal* channels) and *mudflats* for foraging. *Marsh* invasion by *Spartina alterniflora* appears to eliminate some of the elements necessary for sustained rail *populations* that are provided by *Spartina foliosa*.

In the short term, the effects of *tidal marsh* invasion by *Spartina alterniflora* appear to be beneficial to rails. Because it is more fertile and can colonize elevations both higher and lower than the native *Spartina foliosa*, breeding and sheltering habitat are more rapidly provided for rails in restored marshes. However, the long-term impact of *non-native Spartina* invasion on California clapper rails is unknown. Along the Atlantic coast, vast *Spartina alterniflora* marshes provide the primary habitat for the east coast clapper rail (*Rallus lonogirostris crepitans*). However, the east and west coast environments are structurally quite different. If the structure of remaining and restored *tidal* marshes in California approach those of *Spartina*-dominated east coast salt marshes (broad *Spartina* plains with infrequent large *tidal* creeks), the carrying capacity of California estuaries for the clapper rail may become permanently impaired.

Human Disturbance: Clapper rails vary in their sensitivity to human disturbance, both individually and between marshes. Clapper rails have been documented nesting in areas with high levels of disturbance, including areas adjacent to trails, *dikes*, and roads heavily used by pedestrian and vehicular traffic (J. Didonato pers. comm., Baye *in litt.* 2008). In contrast, Albertson (1995) documented a rail abandoning its territory in Laumeister, shortly after a repair crew worked on a nearby transmission tower.

Data on reproductive success of nests near heavily trafficked areas are lacking. Clapper rails nesting next to regularly disturbed areas are likely to be subject to higher rates of predation due to easy access provided by trails, *dikes*, and roads. Disturbance of incubating or brooding adults may translate into reduced hatch or fledge success of young through increased nest predation if the adult vacates the nest, or through temperature stress (heat or cold) due to lack of thermoregulation by the adult. Reduced reproductive success results in reduced recruitment to an already unstable endangered *population*. In addition, continued disturbance may stress the adults and reduce survival through disruption of normal activities, such as reduced foraging or resting time or increased susceptibility to predators. Reduced survival of adult clapper rails may also impact the long-term viability of the *population*, which has been identified as the most critical life stage in *population* models (M. Johnson unpubl. data; Foin *et al.* 1997).

The ramifications of disturbance related to human traffic during breeding season primarily include effects on eggs and chicks or the season's reproductive effort. In addition, anthropogenic noise may also impact survival of adults. Adults may be more responsive to noise during the breeding season, as their mating system is based primarily on auditory signals. Loud noises may elicit calling or prevent advertising calls from being heard, which could disrupt pair bonding and

mating efforts. Studies of noise criteria suggest that noise levels above 80 to 85 decibels (dB) are disruptive to normal behavioral patterns in birds (Transportation Noise Control Center 1997). Clapper rails may be sensitive to noise throughout the year, as rails were heard calling in response to a nearby jackhammer in September (J. Evens *in litt.* 2009). The use of recorded call techniques for censusing California clapper rails may pose an unacceptable threat to at-risk *populations* (B. Grewell *in litt.* 2006b).

Clapper rail reactions to disturbance may vary with season; however, both breeding and non-breeding seasons are critical times. Disturbance during the nonbreeding season may primarily affect survival of adult and subadult rails. Adult clapper rail mortality is greatest during the winter (Eddleman 1989, Albertson 1995), primarily due to predation (Albertson 1995). Human-related disturbance of clapper rails in the winter, particularly during high *tide* and storm events, may increase vulnerability to predators. The presence of people and their pets in the high *marsh* plain or near upland areas during winter high *tides* may prevent rails from leaving the lower *marsh* plain (Evens and Page 1983). Rails that remain in the *marsh* plain during inundation are vulnerable to predation due to minimal vegetative cover available (Evens and Page 1986). For example, during a winter high *tide* at Palo Alto Baylands, a black rail (*Laterallus jamaicensis*) was captured and eaten by a great blue heron (*Ardea herodias*) when the presence of bird watchers standing in and near the only available cover eliminated the rail's escape route (J. Takekawa pers. comm.). This situation is exacerbated in small *diked* marshes with little to no high *tide refugia* or high *marsh* plain.

Although clapper rails may occur in areas with high levels of human-related disturbance, the effects of the disturbance on the rails is unknown and potentially significant. Many marshes only support very small clapper rail *populations* (e.g. only two rails detected at Benicia State Recreation Area in 2005; Herzog *et. al.* 2005), which suggests that even minor incursions could disrupt and potentially extirpate vulnerable small *populations* or *subpopulations*. Because most clapper rail marshes are subjected to a variety of uses, the cumulative detrimental effects may be appreciable. Numerous routine human activities have the potential to adversely affect individual rails and overall *population* viability, for example, flood control; *levee*, dredge lock, pipeline, and powerline maintenance; recreational uses including bird watching; human and domestic animal incursion from adjoining developments; mosquito control ditching, spraying; use of ATVs/Argos in baylands; etc.

Litter: Refuse also affects habitat quality. Although clapper rails often seek refuge on flotsam, during flood *tides* litter of various kinds also supports *populations* of predators such as Norway rats. In some cases the accumulation of litter may kill *marsh* vegetation or be a threat to clapper rail nests. In other cases woody flood debris may provide a structure upon and around which native vegetation may grow (e.g., trellis for *Sarcocornia*), ultimately providing potential nesting opportunities as well as high *tide refugia*. Thus, it is important to distinguish between natural debris and human litter and refuse. Some forms of litter, such as plastic and balloon strings, directly threaten clapper rails, as evidenced a dead clapper rail found tangled in the string of a rubber balloon (Albertson 1995).

Predation—Throughout the bay, the remaining clapper rail *population* is besieged by a suite of mammalian and avian predators. Mammalian species, such as red fox, Norway rats, raccoons,

skunks, and cats, are common terrestrial predators. They are also likely to impact salt *marsh* harvest mice and other native species, such as black rails and *endemic tidalmarsh* song sparrows. Other species, such as gray fox and opossums, are also considered potential predators due to their foraging habits, but their impacts to *tidalmarsh* species are less well documented.

Precipitous declines in South Bay rail *populations* during the mid to late 1980s are attributed largely to intensive predation by the recently arrived red fox (Foerster *et al.* 1990, Albertson 1995). Rail carcasses and egg remains have been found outside of active red fox dens (Foerster and Takekawa 1991). Between 1991 and 1996, a significant negative correlation existed between breeding densities of rails and average fox abundance, such that sites with the highest densities of foxes had no rails. In addition, there was a significant positive relationship between the growth rate of clapper rail *populations* and red fox trapping success in the preceding year. Albertson (1995) suggested that in the South Bay, predation by red foxes posed the most serious threat to adult clapper rails.

Non-native red foxes are present in the North Bay as well as the South Bay (California Department of Fish and Game unpubl. data). Recent preliminary evidence suggests that red foxes in the North Bay (Petaluma, Santa Rosa, and Sebastopol) are *non-native*; however, red foxes from the Montezuma Hills area near the Suisun Bay are genetically more similar to the native Sierra Nevada red fox (*Vulpes vulpes necator*) (B. Sacks *in litt.* 2009). To date, no quantitative data are available on rail mortality due to *non-native* red fox in the North Bay or near Suisun. *Non-native* red fox have been observed since 1988, however, and anecdotal evidence suggests that foxes have been a factor in declines in rail detections at the mouth of Sonoma Creek (Evens 2000a).

Predation consistently takes a high toll on both nest success and hatching success although the impact of predators on clapper rails varies with *marsh*. Chicks and eggs are vulnerable to predation by the entire suite of predators. Norway rats appear to take the majority of eggs lost to predators (Harvey 1988, Foerster *et al.* 1990, Striplen 1992). Foerster *et al.* (1990) found the majority of documented nest losses were due to rats and raccoons. Of 54 active clapper rail nests that contained 348 eggs, predators were responsible for the loss of 115, rodents destroyed 108, foxes destroyed 4, and snakes destroyed 3 (Striplen 1992). An additional 43 eggs failed to hatch due to nest abandonment or inundation, and 38 disappeared during incubation. Estimates of nest predation may be underestimated, however, because certain predators, particularly red fox, are known to carry eggs away from nests prior to consumption. Red fox-depredated rail eggs ($n = 4$) were recovered an average of 5.8 meters (19 feet) from the nest in the South Bay (Striplen 1992). Such displaced eggs may be overlooked by observers, and nest failure mistakenly attributed to other causes, such as adult abandonment or nest inundation. Gopher snakes (*Pituophis melaoleucus*) have taken several clapper rail nests at Laumeister Marsh, and it is possible that ground squirrels and long-tailed weasels (*Mustela frenata*) may take clapper rail nests while foraging in marshes (J. Albertson *in litt.* 2006).

Avian species are also important predators of *tidal marsh* birds and mammals, including clapper rails. *Populations* of many native avian species (common ravens, American crows, California gulls) are artificially increased above historical *population* levels due to the increased availability of food resources and nesting opportunities associated with human activities. Clapper rail

predation from these species has correspondingly been elevated above historical levels. Other species, such as the northern harrier, have been pushed from much of their nearby upland habitat by urban development, and their foraging activities are locally concentrated in the wetland areas. Common ravens and red-tailed hawks are known to nest in electrical towers and buildings and forage in various nearby marshes of South San Francisco Bay (Albertson *in litt.* 2009a). The peregrine falcon is also a likely predator of the clapper rail, and *populations* of this species have increased locally in recent years as a result of peregrine falcon recovery actions.

Landfills and urban areas provide food resources that would otherwise not be available, while buildings, towers, and other human-made structures provide nesting and roosting opportunities. There are four landfills directly adjacent to the Don Edwards San Francisco Bay National Wildlife Refuge (Refuge): Sunnyvale, Palo Alto, Newby Island, and Tri City. Predators of California clapper rail eggs, such as California gulls and common ravens are attracted by these facilities. California gull *populations* have dramatically increased since the 1980s due to the availability of food resources, largely from landfills, coupled with the availability of nesting habitat on dry salt ponds and *levees*. California gull *populations* in the South Bay have increased from fewer than 200 breeding birds in 1982 to over 46,800 in 2008 (Ackerman *et al.* 2009). It is estimated that gulls spend 20 percent of their foraging time at landfills in the South Bay. In a study by Ackerman *et al.* of gull movement in relation to landfills, it was determined that California gulls from a breeding *colony* at pond A6 in the Alviso area of the South Bay arrived at landfills at 6:00 in the morning and left at 6:00 in the evening when the landfills were closed and the exposed refuse was covered (Ackerman *et al.* 2009). Both red-tailed hawks and common ravens, both predators of rail eggs, nest in electrical towers within many marshes, making *tidal marsh* birds and mammals more vulnerable to predation.

Landfills have also been identified as a major source of feral cats on the Refuge, and steps are currently being taken to limit the numbers of cats entering the Refuge from these sites (J. Albertson *in litt.* 2006). In addition, the numerous Bay Area *dikes* and trails allow feral cats easy access to clapper rails, as well as other rare species like California black rail, California least tern, western snowy plover, and salt marsh harvest mouse (American Bird Conservancy 2006). For instance, many sections of the Bay Trail and other public trails have large *populations* of feral cats, many of which are fed daily by well-meaning members of the public or organized cat advocate groups. Also, direct human-caused disturbance to the California clapper rail occurs in these locations of the trail (Albertson *in litt.* 2009b). Five general areas within the scope of this draft recovery plan were identified as sites where cat predation is considered a threat to sensitive bird species: Don Edwards San Francisco Bay National Wildlife Refuge, San Pablo Bay wetlands, Benicia State Recreation Area, Eastshore wetlands (Alameda County), and Elkhorn Slough (Monterey County) (American Bird Conservancy 2006).

Encroaching development not only displaces lower order predators from their natural habitat, but also adversely affects higher order predators, such as coyotes, which would normally limit *population* levels of native and *non-native* predators, especially red foxes (Albertson 1995). This is exacerbated by predator release programs, which relocate nuisance animals from adjacent urban areas. Hunting intensity and efficiency by raptors on clapper rails also is increased by electric power transmission lines and access to boardwalks, which cross *tidal* marshes and provide otherwise-limited hunting perches. Proximity of marshes to urban areas and placement

of shoreline riprap favor rat *populations*, and result in greater predation pressure on clapper rails in certain marshes. These predation impacts are greatly aggravated by a reduction in high *marsh* and natural high *tide* cover in marshes (Sibley 1955, Evens and Page 1986).

The introduced horse mussel may also cause some rail mortality by trapping the bills or feet of birds that have stepped on or probed into the shell (DeGroot 1927). Emaciated rails with mussels clamped onto toes or bills are occasionally observed (U.S. Fish and Wildlife Service unpubl. data).

Contaminants—Environmental contaminants may affect the health and vigor of clapper rails directly through toxic effects to individuals, or indirectly through effects to organisms upon which the rail depends. Acute poisoning associated with oil or toxic material spills could result in rail mortalities within affected habitat. A large oil spill in South Bay marshes could be *catastrophic* for the rail *population*. To date, most direct contaminant impacts to the rail have likely been due to lifetime exposures at chronic, sub-lethal concentrations that alter individual fitness. Known contaminants of concern for rail recovery in the San Francisco Bay Estuary include mercury, selenium, PCBs, and petroleum hydrocarbons. The potential toxicological effects of long term chronic contaminant exposures can include reproductive impairment, compromised immune function, reduced growth, deformity, and altered behavior. While few adult clapper rail mortalities have been directly attributed to contaminants, elevated mercury levels have been found in the tissues of some dead adults. Reproduction in clapper rails has been documented as poor, and contaminants, particularly mercury and perhaps PCBs, are the most likely contributors (Schwarzbach *et al.* 2006).

Contaminants could also indirectly impact rails by altering habitat features such as *benthic* prey density or nesting cover. Petroleum hydrocarbons and trace elements, such as arsenic, copper, silver, cadmium, and lead, may be an indirect hazard through toxicity to *benthic* prey. Although *benthic* organism densities and species composition are known to be altered within the bay by contaminants at some locations (San Francisco Estuary Institute 1999), the effect within rail habitat has not been systematically assessed.

Also of potential concern are newer environmental contaminants that are rarely monitored and poorly understood. Unmonitored contaminants in San Francisco Bay include such chemicals as pharmaceuticals, plasticizers, flame retardants, and detergent additives (San Francisco Estuary Institute 2000). Toxic effects of many of these chemicals to rails and other *estuary* biota are not known. In other species, some of these chemicals have caused endocrine disruption and altered gender development through *in ovo* exposures (Colburn and Clement 1992).

With the exception of the largest deepwater discharges of industry and some municipalities, much of the ongoing contamination of the bay enters at the margins, often through *tidal marsh* habitat. Many, if not most, *tidal* marshland *sediments* are more contaminated than open bay *sediments* (Collins and May 1997). As a omnivore inhabiting the margins of the bay, the clapper rail is exposed to sediment-born contamination of baylands, and may be particularly at risk of exposure to those chemicals that bioaccumulate in *benthic* prey. When comparing diving ducks with other species, the higher concentrations of selenium were found in *benthic* foragers (Ohlendorf *et al.* 1986). Contaminants that are toxic to vertebrates, persist in *sediments*, and

transfer and accumulate in clapper rail prey, present the greatest contaminant hazards to clapper rail recovery.

For the past 75 years or more the greatest densities of breeding rails have been found in marshes of the South Bay (DeGroot 1927, Gill 1979, Harvey 1988). *Freshwater* inflows to the South Bay are substantially more limited than in the North Bay, which receives inflow from the Sacramento and San Joaquin rivers. As a consequence, the residence time for water and also waterborne contaminants is substantially longer in the South Bay. Previous investigators have found a variety of contaminant problems in the South Bay, with silver, mercury, and selenium found to be elevated in bay biota (Luoma and Cloern 1982, Thomson *et al.* 1984, Ohlendorf *et al.* 1986, Smith *et al.* 1986, Luoma and Phillips 1988, Ohlendorf and Fleming 1988, Ohlendorf *et al.* 1991, Lonzarich *et al.* 1992). Mercury and selenium are of particular concern because they are known to accumulate in avian eggs in proportion to the maternal dose, and to adversely impact birds by directly reducing the hatchability of eggs, as well as reducing growth and post-hatch survival of juveniles exposed in the egg.

The following is a brief synopsis of recent contaminant investigations in the San Francisco Bay Estuary. The focus is on contaminants that have been identified as potential hazards to California clapper rails. Mercury, selenium, organochlorine pesticides, polychlorinated biphenyls (PCBs), and petroleum hydrocarbons are discussed. It is important to note that this list is not all-inclusive, and that there are many other compounds being released into the environment that may also adversely affect clapper rails and other *tidal marsh* organisms. Additional details on environmental contaminants in San Francisco Bay are presented in **Appendix E**.

Mercury: Mercury accumulation in eggs is perhaps the most significant contaminant problem affecting clapper rails in San Francisco Bay. California is geologically enriched with mercury, and anthropogenic activities, such as mining for mercury and gold, have released large amounts of mercury in northern California and San Francisco Bay (Schwarzbach *et al.* 2006). Mercury *bioaccumulation* and toxicity to clapper rails are not simple functions of mercury concentration in *sediments*, but depend on rates of methylation that are mediated by bacterial activity and other abiotic factors. Methylmercury concentrations in *tidal marsh sediments* appear to be more variable than total mercury concentrations (U.S. Fish and Wildlife Service unpubl. data). Preliminary results suggest that *sediment* methylmercury concentrations are related to *slough* order, with higher concentrations of methylmercury occurring in higher order channels (San Francisco Estuary Institute 2008).

Mercury is extremely toxic to embryos and has a long biological half-life. Virtually all of the mercury in bird eggs is methylmercury. Toxic effects of mercury in bird eggs have been documented by many investigators in both laboratory and field studies (*e.g.*, Wolfe *et al.* 1998). Fimreite (1971) observed hatchability declines in ring-necked pheasants (*Phasianus colchicus*) when egg concentrations of methylmercury were between 0.5 and 1.5 µg/g, fresh wet weight (fww). In 1992, fifty percent of all the fail-to-hatch California clapper rail eggs from the South Bay were above 0.5 µg/g concentration (fww) and 20 percent of failed eggs in the North Bay were above this concentration (Schwarzbach *et al.* 2006). Twenty-five percent of all the 1992 fail-to-hatch rail eggs were above the 0.86 µg/g (fww) effects threshold estimated for mallards (Heinz 1979).

In 1998 and 1999, a similar study was conducted in the North Bay (U.S. Fish and Wildlife Service unpubl. data). Mercury concentrations in 22 fail-to-hatch eggs ranged from 0.20 to 3.5 µg/g (fww). Concentrations in half of these eggs were above 1.00 µg/g (fww). Mercury concentrations in 5 failed eggs from Hayward Marsh in 1998-99 ranged from 1.28 to 2.12 µg/g (fww). Mercury concentrations in rail eggs appear to vary with position of the nesting territory within a given *marsh*. In addition, maximum methylmercury concentrations in *marsh sediment* were correlated with mean mercury concentrations in failed eggs. In 1998, three embryos from Wildcat Marsh (Contra Costa County) exhibited polydactyly and reduced digits and limbs. Schwarzbach *et al.* (2006) concluded that elevated chromium and barium were among the most likely candidate trace elements responsible for abnormalities; but they could not rule out the possibility that mercury also contributed to the occurrence of deformities.

In summary, three conclusions may be drawn: 1) mercury is accumulated in California clapper rails and deposited in their eggs at potentially embryo toxic concentrations within both the North and South Bay, producing failure in up to one third of clapper rail eggs laid; 2) methylmercury in *sediments* is predictive of the mercury hazard to rail reproduction; and 3) the mercury hazard of North Bay marshes is not less than the South Bay (Schwarzbach *et al.* 2006).

Selenium: The two major potential sources of selenium to the San Francisco Bay Estuary are irrigation drainwater from the San Joaquin River and discharges from the six major oil refineries. Both sources enter the *estuary* in the northern reaches of the bay. Mean selenium levels in the San Francisco Estuary are below the current aquatic life water quality criteria of 5 µg/L (EPA 2009). The Regional Monitoring Program for 1997 (San Francisco Estuary Institute 1999) reported total selenium concentrations throughout the bay from 0.03 to 2.20 µg/L, with highest concentrations detected in the South Bay. Inflows diverted to the Central Valley Project and State Water Project canals usually average about 1 µg/L selenium. However, this single criterion is insufficient to protect aquatic birds from bioaccumulative effects of selenium in aquatic food chains (Stewart *et al.* 2004). San Francisco Bay is considered a selenium-impaired waterbody due to *bioaccumulation* of selenium in biota including *subtidal* clams, sturgeon, and diving ducks (Ohlendorf *et al.* 1986) and has been officially listed by the San Francisco Bay Regional Water Quality Control Board as such under section 303(d) of the Federal Clean Water Act.

Selenium has been considered a contaminant of concern for wildlife in the bay since Ohlendorf *et al.* (1986) documented that selenium concentrations in diving duck livers collected in the South Bay were comparable to concentrations in ducks at Kesterson, where selenium caused embryo deformities in aquatic birds and greatly reduced hatchability of avian eggs. However, the few rail abnormalities found within the bay (Schwarzbach *et al.* 2006) thus far have not been linked to elevated selenium concentrations in eggs.

The *in ovo* threshold for selenium exposure that causes toxic effects on embryos of California clapper rails is unknown. The *in ovo* embryo toxicity threshold for selenium in black-necked stilts (*Himantopus mexicanus*), another *benthic* forager, is 6 µg/g (dry weight; dw) (Skorupa 1998). Clapper rail eggs collected from the North Bay in 1987 contained up to 7.4 µg/g selenium (dw) (Lonzarich *et al.* 1992). Selenium concentrations found in north bay eggs in 1986 were two to three times higher than selenium concentrations in the South Bay. This pattern is

consistent with the fact that major selenium inputs to the *estuary* enter via the North Bay and delta. Investigations of fail-to-hatch clapper rail eggs in the South Bay in 1992, and in the North Bay in 1998, have not duplicated the elevated selenium results of Lonzarich *et al.* (1992). Maximum egg selenium concentrations in more than 60 eggs were less than 3.2 µg/g (dw). It seems unlikely that current selenium concentrations in the bay are having a significant impact on clapper rail reproduction, but that could change if selenium loadings to the *estuary* increase.

Organochlorines: San Francisco Bay has a history of organochlorine contamination from the use of chlorinated hydrocarbon pesticides and polychlorinated biphenyls (PCBs) from the 1950s through 1975 (Venkatesan 1999). Organochlorines are persistent in the environment and are still commonly detected in *sediment* samples throughout the bay (San Francisco Estuary Institute 2000). As a *benthic* forager, rails are exposed to these compounds in *sediment* and through *benthic* organisms. Lonzarich *et al.* (1992) noted a substantial decline in rail egg organochlorines between 1975 and 1986-87 random egg collections. In a follow-up study in 1992, 22 fail-to-hatch clapper rail eggs from the South Bay were analyzed for organochlorines (Schwarzbach *et al.* 2001). Results from these eggs showed a continuing trend of decline in organochlorine concentration. Neither the 1986 random egg collections nor the 1992 failed egg collections found a reduction in clapper rail eggshell thickness. Organochlorine concentrations in failed clapper rail eggs collected in 1998-1999 from two Central Bay marshes were similar to those from the South Bay in 1992 (Schwarzbach *et al.* 2006). These studies concluded that organochlorine pesticide concentrations were not likely to cause adverse effects on clapper rail reproduction (Lonzarich *et al.* 1992, Schwarzbach *et al.* 2001, Schwarzbach *et al.* 2006).

Trends in PCB concentrations in eggs differed from those of organochlorine pesticides. PCB concentrations declined from an average of 2.86 µg/g (fww) in 1975 to 0.82 µg/g (fww) in 1986-1987 (Lonzarich *et al.* 1992). In contrast, PCB concentrations in rail eggs collected from the South Bay in 1992 averaged 1.30 µg/g (Schwarzbach *et al.* 2001). The general trend of decreasing PCB concentrations continued in eggs collected in 1998-1999 from the Central Bay (Schwarzbach *et al.* 2006), with an average of 0.56 µg/g (fww). It is interesting to note that in each year during which clapper rail eggs were collected, PCB concentrations were greater than concentrations of any other organochlorine pesticide quantified, however, only one collected had PCBs high enough to have impacted hatchability (Schwarzbach *et al.* 2006).

Toxicity of PCB to clapper rail embryos is unknown, while toxicity of PCB congeners to avian embryos varies greatly. Decreased hatching success in white leghorn chickens (*Gallus domesticus*), the most sensitive avian species tested, was associated with PCB egg residues of 0.87 µg/g (ww) in a feeding study with Aroclor 1242 (Britton and Huston 1973, Schwarzbach *et al.* 2006). Of the 1992 rail eggs, 18 of 22 contained PCB concentrations above this threshold. If rails are as sensitive to PCB toxicity as chickens, they may be at risk from PCBs.

Petroleum hydrocarbons: San Francisco Bay Estuary has many potential sources of petroleum hydrocarbon release, as it is highly urbanized, with six oil refineries, substantial ship and oil tanker traffic, and a large number of gas-powered vehicles. As a result, petroleum hydrocarbons are commonly detected in bay waters and *sediment*. Polycyclic aromatic hydrocarbons (PAHs) are among the most toxic hydrocarbons; many are carcinogenic or mutagenic (Eisler 1987).

Rails may be exposed to petroleum hydrocarbons both internally through normal foraging and externally from an oil spill.

There have been several major oil spills within San Francisco Bay in the last decades, including long-term leaks from the SS *Jacob Luckenbach* along the northern California coast since 1953; the Martinez Manufacturing Complex of Shell Oil Company, Peyton Slough, California, 1988; Tosco Corporation Avon Refinery spill, Martinez, California 1980; the Cape Mohican oil spill, San Francisco, 1996; chronic releases by Chevron from Castro Cove near Richmond, Contra Costa County; the Kinder-Morgan Suisun Marsh oil spill of 70,000 gallons from a pipeline rupture in April 2004; and a major spill of 58,000 gallons of oil from the *Cosco Busan* in San Francisco Bay, November 2007 (U.S. Fish and Wildlife Service 1997b, California Department of Fish and Game 2009). These spills were due to a number of causes including shipping accidents, a pipeline rupture and an open valve at a refinery, leaks from a sunken ship, etc. Many of the spills affected the interior shoreline of the bay, with impacts to the Central Bay and Carquinez Strait. Numerous marshes in both areas support clapper rails. Although no clapper rails were identified in salvage or cleanup operations, rails may have been oiled and escaped detection due to their normally secretive behavior. The effects of an oil spill depend on the degree of oiling and the nature and weathering of the oil. A large oil spill in the South Bay, where clapper rail *populations* are more densely concentrated, could have serious ramifications for the long-term survival of the species.

Harvest—Overharvesting by commercial and sport hunting between 1850 and 1913 initially contributed to the depletion of the California clapper rail *population*. Thousands of rails were reported killed in a single day in 1859 (Wilbur and Tomlinson 1976). Up to 5,000 rails of several species were reported killed during a one-week period in 1897 in south San Francisco Bay (Gill 1979). Between 1890 and 1910, reports of 30 to 50 rails taken a day were not uncommon. After the enactment of the Migratory Bird Treaty Act in 1918 brought about the cessation of hunting, rails increased in abundance in the remaining San Francisco Bay Marshes (Bryant 1915, Grinnell and Miller 1944).

e. Salt marsh harvest mouse (*Reithrodontomys raviventris*)

1) Brief Overview

The salt marsh harvest mouse (*Reithrodontomys raviventris*) was listed as a Federal Endangered Species (U.S. Fish and Wildlife Service 1970) on October 13, 1970, and a California State Endangered Species in 1971 (California Department of Fish and Game 2005). It has a recovery priority number of 2C, based on a high degree of threat, a high potential of recovery, and its taxonomic standing as a species. The additional “C” ranking indicates some degree of conflict between the conservation needs of the species and economic development (U.S. Fish and Wildlife Service 1983). A previous recovery plan was written for the species in 1984 (U.S. Fish and Wildlife Service 1984). There are two subspecies: the northern salt marsh harvest mouse (*Reithrodontomys raviventris halicoetes*) lives in the marshes of the San Pablo and Suisun bays,

and the salt marsh harvest mouse (*Reithrodontomys raviventris raviventris*) is found in the marshes of Corte Madera, Richmond, and South San Francisco Bay.

The salt marsh harvest mouse is generally restricted to saline or *subsaline marsh* habitats around the San Francisco Bay Estuary, and is found in mixed saline/*brackish* areas in the Suisun Bay area. Habitat loss due to human actions is the greatest threat to the salt marsh harvest mouse. Habitat loss that threatens salt marsh harvest mouse is due to filling, diking, *subsidence*, changes in water *salinity*, *non-native* species invasions, sea level rise associated with global climate change and pollution. In addition, habitat suitability of many marshes is further limited by small size, fragmentation, and lack of other vital features such as sufficient *escape habitat*. Larger tracts of high quality habitat are needed to maintain stable *populations* over time.

2) *Description and Taxonomy*

Description. The salt marsh harvest mouse (*Reithrodontomys raviventris*) is a rodent (Order Rodentia) in the family Muridae (subfamily Sigmodontinae; **Figure II-11**). The scientific name *Reithrodontomys raviventris* means “grooved-toothed mouse with a red belly.” Both subspecies of salt marsh harvest mouse have grooved upper front teeth, but only a few *populations* of the southern subspecies have animals with a cinnamon- or rufous-colored belly. Both subspecies have rich dorsal brown hair and a unicolored to moderately bicolored tail. The combined head and body length is approximately 7.6 centimeters (3 inches) with an average weight of less than 10 grams (0.353 ounce).

The salt marsh harvest mouse is morphologically similar to the more widespread western harvest mouse (*Reithrodontomys megalotis*), which co-occurs in some habitats. The underside of the western harvest mouse, including its tail, ranges from white to dark gray (Shellhammer 1984). Accurate field identification of mice in salt *marsh* habitats requires special expertise as some *populations* of the salt marsh harvest mouse may exhibit morphological characteristics similar to those of the western harvest mouse, especially in the northern reaches of the *estuary*. Comprehensive morphological comparisons of harvest mouse *populations* in the region are given by Fisler (1965; see **Table II-5**); modifications of those traits for use in field identification are found in Shellhammer (1984). Villablanca and Brown prepared an interim report on the use of molecular and morphological tools to determine if salt marsh harvest mice and western harvest mice are hybridizing as well as to distinguish them by morphological traits (Villablanca and Brown *in litt.* 2004). Results of the study indicate that the two species are not likely hybridizing and that tail length is the most distinguishing character between salt marsh harvest mice (tail lengths of 77.5 mm and greater) and western harvest mice (tail length of 77.4 mm and less).



FIGURE II-11. Salt marsh harvest mouse (Kendal Morris/USFWS)

Table II-5. Key Field characters distinguishing between the salt marsh harvest mouse and western harvest mouse (*adapted from Fisler 1965, Shellhammer 1984*).

Trait	Salt marsh harvest mouse (<i>R. r. raviventris</i>)	Northern salt marsh harvest mouse (<i>R. r. halicoetes</i>)	Western harvest mouse (<i>R. r. megalotis</i>)
tail thickness (20 millimeters from body)	2.1 to 3.0 millimeters (0.083 to 0.118 inch)	2.1 to 3.0 millimeters (0.083 to 0.118 inch)	1.9 to 2.0 millimeters (0.075 to 0.079 inch)
venter (belly) hair color	rusty-cinnamon	white	white
tail hair color	unicolor or indistinctly bicolor (typical)	unicolor or indistinctly bicolor (typical)	distinctly bicolor (typically white hairs below)
average tail:body ratio	94.7 to 105.3	107.0 to 116.8	103.1 to 110.8
tail tip	heavy, relatively blunt	heavy, relatively blunt	relatively pointed
pelage (coat)	relatively thick; long hairs	relatively thick; long hairs	relatively thin; short hairs
activity (during trap, release observation)	relatively placid; infrequent aggressive behavior	relatively placid; infrequent aggressive behavior	relatively active, typical, frequent aggressive behavior
early morning activity	becomes torpid when cold	no torpidity	no torpidity

Taxonomy. The two subspecies of salt marsh harvest mouse were originally described as two distinct species. The type is the salt marsh harvest mouse of San Francisco Bay, *Reithrodontomys raviventris raviventris*, described by Dixon (1908) from Redwood City, San Mateo County, California. Some individuals of the southern subspecies may have a rusty or cinnamon brown belly, although there is variation in this trait among *populations* and many

populations of the southern subspecies have few to no individuals with red bellies. The northern subspecies of San Pablo Bay and the Suisun Marsh area, *Reithrodontomys raviventris halicoetes*, was described from specimens taken in the Petaluma Marsh, Sonoma County, by Dixon (1909). It has a whitish belly, and is overall more similar in appearance to the western harvest mouse. Artificial breeding experiments that attempted to hybridize the two subspecies resulted in low mating success and one litter that was destroyed by the mother (Fisler 1965). This suggests that the subspecies boundary is based on reproductive, as well as geographic, isolating mechanisms.

Despite similarities that led Hooper (1944), Fisler (1965), and others to infer that the salt marsh harvest mouse was derived from an ancestor of the western harvest mouse, *genetic* analysis does not support a close ancestral relationship between the two (Hood *et al.* 1984, Nelson *et al.* 1984, Bell *et al.* 2001, Arellano *et al.* 2005). Instead, *genetic* data suggest that the salt marsh harvest mouse is most closely related to the plains harvest mouse (*Reithrodontomys montanus*), a western interior species that does not occur near the central California coast today.

3) Population Trends and Distribution

Historical Distribution. By the time the salt marsh harvest mouse was distinguished as a species in 1908, extensive *tidal* marshes throughout its range had already been reclaimed for agriculture, salt ponds, and urban development. Therefore, there are no historical records of its abundance or distribution in the *estuary* to use as a baseline.

The salt marsh harvest mouse probably occupied most of the middle tidal, or *Sarcocornia*-dominated, *marsh* plains and high *marsh* zones of San Francisco Bay, San Pablo Bay, and the Suisun Marsh prior to the significant *marsh* reclamation of the 1840s. Although estimates of historic *tidal marsh* area in the San Francisco Bay Estuary are not precise enough to distinguish between suitable and unsuitable habitats for the salt marsh harvest mouse, most of the mature *tidal* marshes in the region had extensive middle *marsh* plains and even more extensive high marshes. It is likely that most suitable habitat supported salt marsh harvest mice, since the species can colonize rapidly under favorable conditions (Geissel *et al.* 1988, Bias and Morrison 1999), and habitats were naturally contiguous and extensive. Thus, the area inhabited by the salt marsh harvest mouse prior to *tidal marsh* reclamation could have approached 77,000 hectares (190,000 acres), the total *tidal marsh* area (Dedrick 1989, Goals Project 1999).

Current Distribution. The current distribution of the salt marsh harvest mouse can be found in **Figure II-12**. Distribution can be estimated from the remaining suitable *diked* and *tidal marsh* habitat, and the review of live-trapping surveys, although trapping data are limited (Shellhammer 1984, Zetterquist 1976, Larkin 1984, Bias and Morrison 1993). Much of the data on local abundance and distribution of the salt marsh harvest mouse have been derived from local short-term studies, usually conducted on privately owned *diked* baylands proposed for land use changes (H. Shellhammer pers. comm. 2005). These data must be interpreted with caution as data become quickly outdated.

Southern subspecies population trends

The *population* status of the southern subspecies is more precarious than that of the northern subspecies. Few major, resilient, or secure *populations* persist (Roberts Landing, Hayward Marsh, Baumberg, Mayhews Landing, Calaveras Point Marsh, New Chicago Marsh, Renzel/ITT Marsh, Redwood Shores, in addition to likely *populations* at Bair Island, Greco Island, Mowry Slough, and other sites). These were very small and isolated compared with the historical pattern of distribution and abundance of the subspecies. All major *population* centers of the southern subspecies are remote from one another based on dispersal distances known for the species. The small *populations* and higher degree of isolation of the southern subspecies in Marin County indicate a high probability of local extirpation due to inability to recolonize following local extinction.

Although salt marsh harvest mouse abundance does not appear to correspond with the distribution of its native *tidal salt marsh* due to the relatively common occurrence of the species in areas of nontidal or *microtidal Sarcocornia pacifica* (pickleweed) *marsh*, this appears to be an artifact of surveying effort. A fairly small fraction of large pure *tidal marsh* has been surveyed for the species, while a large fraction of *diked* marshes have been surveyed. The few large *tidal* marshes that have been surveyed have yielded very high densities of the mouse (Duke pers. comm. 2005).

Studies by Shellhammer (unpubl. data) indicate that *population* size is generally correlated with the depth of the *Sarcocornia* plain (*i.e.*, the middle zone of *tidal* marshes). There are indications that deep (from shore to bay) *Sarcocornia* marshes, especially if they have islands of *Grindelia* within them, may provide enough habitat for the mice such that they can compensate for extremely narrow high marshes at their upper edges. *Corridors* (sometimes referred to as strip marshes) tend to have narrower *Sarcocornia* zones (as well as extremely narrow high *marsh* zones) and support few to no mice. In fact, the narrower the strip *marsh*, the more frequently and intensely it floods (Albertson *in litt.* 2009a). Most of the marshes of the South San Francisco Bay are strip-like marshes and, as such, support few mice. In strip-like marshes identified as *marsh corridors* to connect habitat areas, the relative value of the width and complexity of the high *marsh* zone increases as the width of the middle *marsh*, or pickleweed/*Sarcocornia* zone, diminishes (Shellhammer unpubl. research).

Northern subspecies population trends

The fringing salt marshes along northern San Pablo Bay (Petaluma River to Mare Island Strait) support the largest *population* of the northern subspecies of salt marsh harvest mice in San Pablo Bay. Outside of the Highway 37/Mare Island Marsh there are other major centers of stable or large *populations*. These include *diked* salt marshes south of Black John Slough (lower Petaluma River) and tidal/*microtidal* marshes around Gallinas Creek, Coon Island, Fagan Marsh, and Point Edith to Middle Point. Patchy, diffuse, and unstable *populations* of large cumulative size occupy Suisun Marsh.

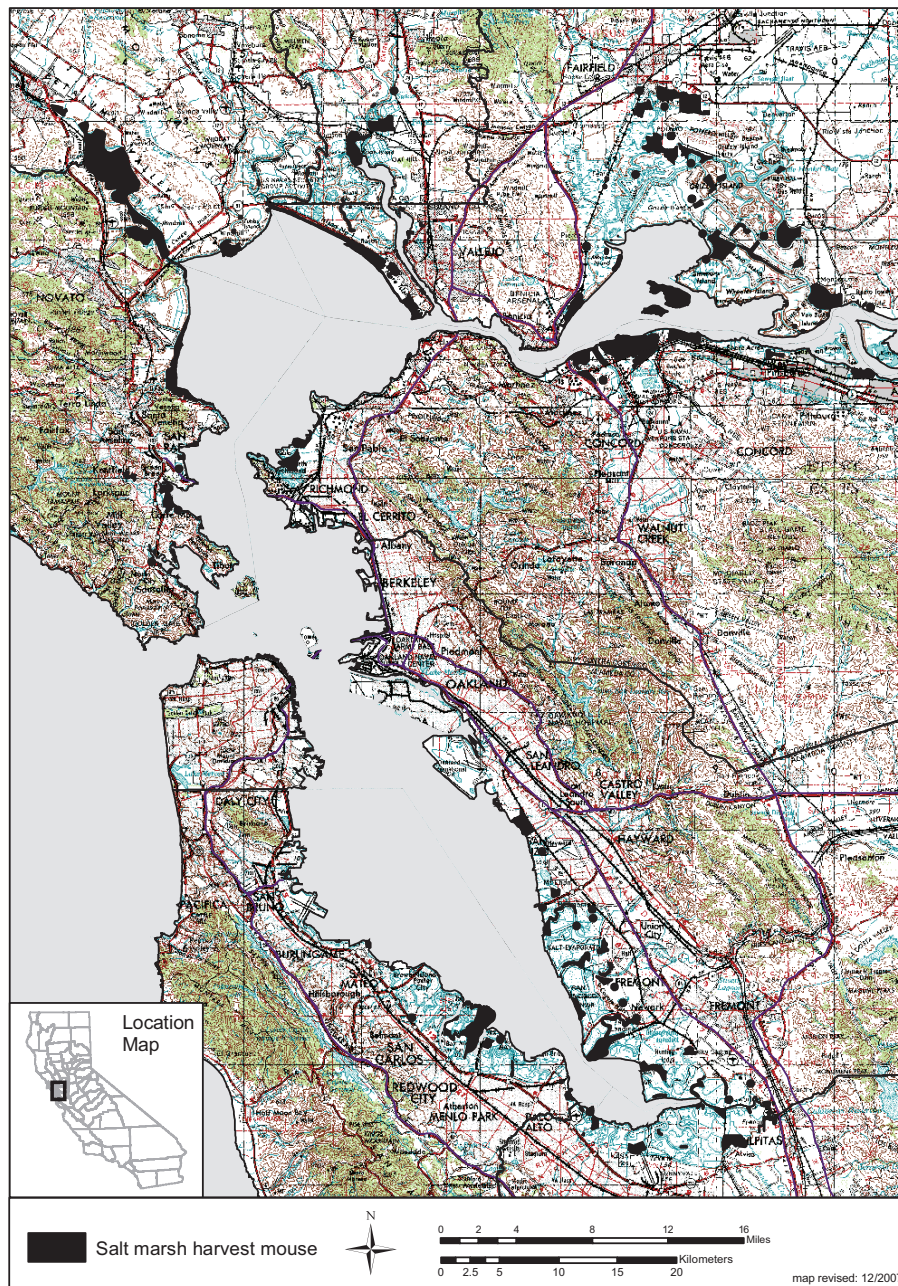


Figure II-12.

Figure II-12. Distribution of salt marsh harvest mouse

The northern subspecies is more widespread and patchy in distribution in both *diked* and *tidal* marshes than the southern subspecies, although densities may be very low outside of the Highway 37 and Mare Island Marshes and marshes of the Contra Costa County shoreline. Like the southern subspecies, many northern subspecies *populations* have been displaced from *tidal* marshes to unstable *diked Sarcocornia* marshes. Most of the *populations* in *diked* managed marshes of Suisun Marsh appear to depend on opportunistic colonization of unstable *Sarcocornia* vegetation. Salt marsh harvest mice may become abundant in portions of *diked brackish* marshes, especially in “mouse preserves” where extensive tall dense cover of *Sarcocornia* vegetation and various species of *brackish* vegetation develop (Finrock 2000) because of effective and consistent water management. Unstable, unmanaged, or poorly managed *diked Sarcocornia* marshes, however, are highly vulnerable to *catastrophic* flooding and local extirpation. Salt *marsh* harvest mice are sometimes also found in significant numbers in grasslands at the upper edge of *diked* marshes around San Francisco Bay (Zetterquist 1976, Shellhammer *et al.* 1982, Johnson and Shellhammer 1988, Shellhammer *et al.* 1988, Thompson *in litt.* 2009), as described below under Habitat Characteristics/Ecosystem. The extent to which this habitat is utilized is not clear.

Less *population* survey information is available for the northern subspecies, despite its larger range, than for the southern subspecies.

4) *Life History and Ecology*

Reproduction. Male salt marsh harvest mice are generally sexually active from April through September, while the female breeding season extends from March through November for the northern subspecies, and May through November for the southern subspecies (Fisler 1965). Bias and Morrison (1993) suggest that the breeding season of the Mare Island *population* (northern subspecies) extends from August through November; more than 30 percent of the females trapped were pregnant during September and October. Compared with environmentally determined mortality factors, reproduction does not appear to be a limiting factor for the species.

Home range. Telemetry studies of the northern salt marsh harvest mouse at Mare Island Marshes found a mean home range size of 0.21 hectare (0.52 acre), and a mean linear distance moved of 11.9 meters (39 feet) in 2 hours (Bias and Morrison 1999). Most movements occurred in June, and least in November. Mare Island mean home ranges were much larger than those estimated by Geissel *et al.* (1988) for the southern subspecies, which were no greater than 0.15 hectare (0.37 acre). Movements through open habitats were not restricted to rare or extraordinary events (Kovach and Pomeroy 1989; Geissel *et al.* 1988; Bias and Morrison 1993, 1999).

Competition. *Population* dynamics based on interactions between harvest mice and other small mammals are not well understood (Blaustein 1980, Geissel *et al.* 1988, Bias and Morrison 1993, Bias 1994, Wertz-Koerner 1997, Hulst 2000). Hypotheses of *competitive exclusion* in salt marsh harvest mouse *populations*, based on analogy with studies on voles (*Microtus californicus*) and western harvest mice, should be applied with caution to salt marsh harvest mice (Blaustein 1980, 1981; Heske *et al.* 1984).

Predation. Very little is known about predation impacts to the species, although predation related to flooding has been viewed as an important factor (Johnston 1957, Fisler 1965). During high winter *tides* it is common to see great blue herons, great egrets, snowy egrets, ring-billed gulls, California gulls, and American kestrels all taking small mammals from the upper edges and flooded areas of marshes. Protection from predators depends on the dense vegetation cover of typical salt marsh harvest mouse habitat. Mice that leave this cover, or those forced out by flooding, are exposed to predation by hawks and gulls by day, and short-eared owls (*Asio flammeus*) at night (Fisler 1965). Abundant white-tailed kites (*Elanus caeruleus*) and northern harriers (*Circus cyaneus*) frequently forage over thickly vegetated *diked* and *tidal Sarcocornia* marshes in San Pablo Bay during all *tidal* stages (P. Baye pers. observ.), but their impact on salt marsh harvest mice is unknown. Clapper rails (*Rallus longirostris obsoletus*) and herons also occasionally take small mammals (Terres 1980, Josselyn 1983, Meanley 1985). The impact of terrestrial predators on salt marsh harvest mice has not been studied. Potential terrestrial predators include red fox (*Vulpes vulpes*), gray fox (*Urocyon cinereoargenteus*), feral cats (*Felix domestica*), skunks (*Mephitis mephitis*), and raccoons (*Procyon lotor*). Other than predation of exposed mice during *marsh* flooding events, predation is presumably greatest in habitats with incomplete or sparse cover, such as *diked* baylands with patchy vegetation and high proportions of *annual* grasses. These habitats also are usually closer to urban edges where terrestrial predators, such as feral cats, occur. The overall impact of non-flood predation on the recovery of salt marsh harvest mice is less significant than other factors such as habitat quality and size.

5) *Habitat Characteristics/Ecosystem*

The basic habitat of the salt marsh harvest mouse is *Sarcocornia*-dominated vegetation (Dixon 1908, Fisler 1965). Other highly important habitat considerations include high *tide/flood refugia* of emergent *Grindelia* (gumplant; both at the upper edge of the *marsh* and within mature marshes, even at the highest high *tides*), seasonal use of terrestrial grassland, exploitation of suboptimal habitats, and habitat selection in *brackish marsh* vegetation where *Sarcocornia* is a relatively minor component, as often is the case in Suisun Bay marshes.

Salt marsh harvest mice are typically associated with tall, dense, continuous stands of *Sarcocornia pacifica* in saline soil. These stands remain mostly unsubmerged during periods of flooding, or are mixed with other unsubmerged sources of cover, such as taller vegetation (*Grindelia* or debris; Fisler 1965, Rice 1974, Johnson and Shellhammer 1988, Shellhammer *et al.* 1988, Bias and Morrison 1993, Hulst 2000). Within *Sarcocornia* marshes the taller, denser stands tend to support the most salt marsh harvest mice, although they may also be abundant in *tidal* marshes with relatively short *Sarcocornia* canopies. A *Sarcocornia* canopy height of approximately 15 centimeters (6 inches) appears to be the lowest commonly used by salt marsh harvest mice (Shellhammer *et al.* 1982, Fisler 1965). The relationship between *Sarcocornia* height and salt marsh harvest mice abundance may depend on degree of canopy submergence rather than height alone.

The ecological basis for the salt marsh harvest mouse affinity for *Sarcocornia* habitat is probably due to several factors, including year-round cover from predators, use of *Sarcocornia* as a food source, competition with other small mammals, and escape from flooding (Fisler 1965; Shellhammer *et al.* 1982, 1988; Geissel *et al.* 1988, Bias and Morrison 1993). These factors are

not uniquely associated with *Sarcocornia*, however, and there is significant variation in vegetation types used by salt marsh harvest mice. Saline to *subsaline marsh* that lacks *Sarcocornia*, or supports it as a minor component, may be used as habitat by significant numbers of salt marsh harvest mice; this is especially the case in many parts of the Suisun Bay (Botti *et al.* 1986, California Department of Water Resources *in litt.* 2007). There is no trapping evidence to indicate that *Spartina foliosa* (Pacific cordgrass), some *Scirpus spp.* (bulrush, tule), and *Typha* (cattail) vegetation are more than marginal and incidental habitat for the salt marsh harvest mouse (Fisler 1965, Shellhammer *et al.* 1982), although recent studies (2000-2005) in the Grizzly Island and Hill Slough areas within the Suisun Marsh indicate a much greater use of various *Scirpus* species than found in other portions of the range (California Department of Water Resources *in litt.* 2007).

In fact, Shellhammer *et al.* (1982) concluded that mixed stands of native salt *marsh* vegetation dominated by *Sarcocornia* have higher habitat value than pure stands. Salt *marsh* plants suggested as beneficial in mixed stands include *Frankenia salina* (alkali-heath), *Atriplex triangularis* (sparscale), and possibly small amounts of *Distichlis spicata* (saltgrass). The Mare Island *Sarcocornia* marshes are very low in vascular plant species diversity other than *Sarcocornia* and *Cuscuta salina* (parasitic dodder), but support exceptionally tall, dense *Sarcocornia* vegetation and an abundance of salt marsh harvest mice (Bias and Morrison 1993). Although salt marsh harvest mice have a high affinity for the *annual salt marsh* forb *Atriplex triangularis*, due to the inherent winter dieback of this species it has no significant winter habitat value (Rice 1974, Botti *et al.* 1986).

Salt marsh harvest mice commonly occur in the upper portions of salt marshes where terrestrial grasses are absent or remote, while western harvest mice tend to be dependent on proximity to terrestrial grass vegetation (Fisler 1965). However, salt marsh harvest mice frequently utilize terrestrial grassland habitats adjacent to salt *marsh* and grass-*Sarcocornia* *ecotones* (Zetterquist 1976, Shellhammer *et al.* 1982, Johnson and Shellhammer 1988, Shellhammer *et al.* 1988), and this use is highest in the late spring and early summer. Salt marsh harvest mice in eastern San Pablo Bay and Suisun Marsh (northern subspecies) appear to be more widespread in terrestrial grasslands and grassland-*brackish marsh ecotones* than those in the South San Francisco Bay (southern subspecies). Persistent low numbers of salt marsh harvest mice were found in predominantly grassland vegetation at Cullinan Ranch, which is adjacent to Mare Island Marsh, one of the most densely populated habitats of the species (Wertz-Koerner 1997, Hulst 2000). There are few data available on the distance that salt marsh harvest mice are likely to travel from salt *marsh* into terrestrial grassland. Johnson and Shellhammer (1988) speculated that dispersal to grasslands may be driven by competition from California meadow voles, but this has not been consistently shown (Bias and Morrison 1993, Hulst 2000). The use of grasslands by salt marsh harvest mice in the spring has been interpreted as an opportunistic exploitation of a seasonally available resource, rather than use of an essential habitat (Fisler 1965, Johnson and Shellhammer 1988).

The extent to which salt marsh harvest mice used, or would use, native grasslands has not been investigated (Baye *et al.* 2000, Holstein 2000). Native grasses occur infrequently, but in local abundance, along the edges of *tidal salt* and *brackish* marshes in San Pablo Bay and the Suisun Marsh area. Cover is a limiting factor for the northern subspecies (Fisler 1965), and native

Leymus triticoides (wildrye) stands, which provide tall dense cover at all times of the year (P. Baye pers. observ.), may form a better marginal grassland habitat than *annual* European grasses.

Lepidium latifolium (perennial pepperweed) readily invades *brackish* middle *marsh* plains that support significant proportions of *Sarcocornia* vegetation and associated native salt *marsh* plants. It can overtop and shade a *Sarcocornia* understory, and displace all other *tidal brackish marsh* vegetation (P. Baye pers. observ. 1990-2000). *Lepidium latifolium* can form dense, often monotypic stands in high *tidal marsh* zones and terrestrial *ecotones*. It is not known whether or how salt marsh harvest mice use perennial *Lepidium latifolium*. Despite the great and increasing extent of *Lepidium latifolium* in *brackish tidal* marshes historically occupied by salt marsh harvest mice, there have been no quantitative investigations of this relationship.

Studies have documented ecologically significant numbers of salt marsh harvest mice in what have been termed marginal, atypical, and suboptimal habitats (Botti et al. 1986, Geissel et al. 1988, Wertz-Koerner 1997, Hulst 2000). For that reason, it is important to avoid sampling bias caused by locating survey lines only in stands of vegetation determined to be optimum habitat or those thought most likely to produce trap success. This practice ensures failure to identify atypical or suboptimal stands of vegetation that support ecologically significant *populations* of salt marsh harvest mice (Baye 2000, Baye et al. 2000). Very few studies have been conducted on the *marsh* plain in broad *tidal* salt marshes. This makes it difficult to comparatively assess *population* densities, and thereby the importance, of these *tidal* marshes. The few examples that exist (Calaveras Point, Highway 37 marshes) yield significantly high numbers of captures (Duke pers. comm. 2005).

Flood and tidal refugia. Flooding as a factor in habitat quality for salt marsh harvest mice is closely related to vegetation and *marsh* structure. Flooding that submerges vegetation of the middle *marsh* plain may occur from very high *tides* near the summer and winter solstices, *storm surges*, and extreme river outflows into the *estuary*. Fisler (1965) concluded that the January and December *tides* were critical high *tides* that could endanger whole *populations* of salt marsh harvest mice. Prolonged flooding exposes salt marsh harvest mice to predators, and increases the risk of mortality due to exposure or drowning. Although salt marsh harvest mice float and swim well (Fisler 1965), and cross open water without being forced by flooding (Geissel et al. 1988, Bias and Morrison 1999), they do not swim as well as other small salt *marsh* mammals, nor do they dive (Johnston 1957). Mice move locally from flooded salt *marsh* to emergent high ground or vegetation. Salt marsh harvest mice likely remain in their home ranges during high *tide* immersion of *marsh* vegetation, and swim or cling to taller emergent portions of vegetation or floating debris (Johnston 1957, Hadaway and Newman 1971).

The relative importance of landward *marsh* edges as flood *refugia* for salt marsh harvest mice probably differs between narrow and deep *tidal* marshes. Flood *refugia* at landward *marsh* edges appear more important in narrow marshes where mice are concentrated during high *tide* and slightly less important in deeper marshes, given their intramarsh *refugia*. Even in deep *marsh* plains, the only available *refugia* are *Grindelia* vegetation, natural *berms* and *levees*, and trapped floating woody debris along *marsh* edges at creek banks (Johnston 1957; Hadaway and Newman 1971; Bias and Morrison 1993, 1999).

Salinity. *Salinity* may influence salt marsh harvest mouse habitat independent of its correlation with *Sarcocornia*. Zetterquist (1978) found that salt marsh harvest mice were most abundant in portions of *diked* salt marshes where *salinity* was extremely high. A high physiological tolerance for salt in food and water (Fisler 1965, Coulombe 1970) may confer a competitive advantage to salt marsh harvest mouse in harshly saline *marsh* habitats, particularly where competition with more aggressive, but less salt-tolerant, California voles occurs (Geissel *et al.* 1988; Blaustein 1980, 1981). This suggests that otherwise suboptimal *hypersaline* salt *marsh* vegetation and *salt pans* may provide important habitat exploited intermittently by salt marsh harvest mice to cope with interspecific competition. However, this conclusion is uncertain. The wide high *tidal* salt *marsh* plain at Mare Island Marsh consists of nearly pure stands of extremely tall, dense *Sarcocornia* with few local *pans* that are *brackish* for most of the year (P. Baye pers. observ.), yet this *marsh* supports consistently high *populations* of salt marsh harvest mice that coexist with California voles (Kovach and Pomeroy 1989; Bias and Morrison 1993, 1999). Similarly, many tall, dense stands of *Sarcocornia* non-*tidal* seasonal wetlands grow in non-saline to *subsaline* soils (Kovach and Pomeroy 1989, P. Baye pers. observ.).

6) Critical Habitat

No critical habitat has been designated for the salt marsh harvest mouse.

7) Reasons for Decline and Threats to Survival

Most species covered in this draft recovery plan are threatened by similar factors because they occupy the same *tidal marsh* ecosystem. These general threats, faced by all covered species, are discussed in greater detail in the Introduction section of this draft recovery plan (section I.D.). Specific threats to salt *marsh* harvest mouse are described below.

The most fundamental reason for the decline of the salt marsh harvest mouse is loss of habitat through filling (*i.e.*, destruction), *subsidence*, and vegetation change (U.S. Fish and Wildlife Service 1984, Bias and Morrison 1993, Shellhammer 2000). The high and middle, or *Sarcocornia*, zones, of *tidal* marshes have been the most affected. Shellhammer (unpubl. research) has found that the high *marsh* zone, once kilometers deep (from shore to bay) throughout the South San Francisco Bay, is now an interrupted band approximately 2 meters (6 feet) deep. The same study found that the adjacent upland edge (*i.e.*, the *ecotone* between *marsh* and upland) exists today in only 2.5 percent of the South Bay's edge. Habitat losses include areas associated primarily with historical diking and reclamation of *tidal* salt marshes, urban development of *diked* salt marshes, and adverse water management in *diked* *brackish* marshes of Suisun Marsh (Suisun Ecological Workgroup 2001). Other large net losses of nontidal occupied habitat have occurred since the publication of the first recovery plan including 1) grading and development of saline seasonal *marsh* adjacent to Mayhews Landing along old Jarvis Avenue in Newark; 2) re-emergent *Sarcocornia* in subsided, filled *diked* baylands at the residential Redwood Shores development; 3) replacement of *Sarcocornia* with annual seasonal wetland forbs at the Gentry-Pierce site in Fairfield; and 4) large-scale conversion of *Sarcocornia* to seasonal waterfowl habitat through improvements in Suisun Marsh duck clubs.

Significant habitat degradation has continued in some portions of the salt marsh harvest mouse range. Ongoing high-magnitude wastewater discharges from sewage treatment operations and channelized urban runoff into *tidal sloughs* from San Jose to Milpitas (Guadalupe, Alviso, Artesian/Mallard Sloughs, Coyote Creek) have concentrated impacts on fringing *tidal* marshes. The *perennial* depression of channel water *salinity* during high *freshwater* flows has caused conversion of middle *tidal* marsh plains from salt *marsh* to *brackish marsh* dominated by species with very low or negative habitat value to the salt *marsh* harvest mouse (*Scirpus maritimus* [alkali bulrush], *Lepidium latifolium*), and reduced *marsh salinity* (H.T. Harvey and Associates 1997). During years of high rainfall, cumulative *brackish marsh* conversion problems are most severe, although high background *freshwater* outflows may mask the impact of wastewater discharges on *brackish marsh* conversion. As human *population* size and water use increases in the Santa Clara Valley, this problem may worsen.

Extirpated populations may fail to re-establish despite regeneration of suitable habitat conditions, possibly because of constraints on dispersal from source *populations*. Where few widely spaced source *populations* are separated by significant geographic or ecological barriers, there is little chance for recolonization by vagrant *founders*. Many narrow strip-like marshes are the only potential *corridors* between existing larger marshes. Narrow marshes (*i.e.*, those with shallow *Sarcocornia* marsh plains and very narrow high *marsh* zones) are highly unlikely to be functional *corridors*. Marshes of this type that are only 9 to 12 meters (30 to 40 feet) long may be *genetic* and migration “filters”; those longer than that are likely to be complete barriers (Shellhammer unpubl. research).

Flooding of salt marsh harvest mouse habitat in *diked* baylands is influenced by (1) the degree of *subsidence* below sea level, (2) the efficiency of tidegate drains and drainage ditches operating at low *tide*, and (3) the magnitude of flooding. Average rainfall seldom causes complete or widespread submergence of *Sarcocornia* canopies. Extremely high rainfall, managed intake of bay water, overtopping, and *dike* breaching all can completely submerge *Sarcocornia* canopies, and cause mass mortality and dispersal of salt marsh harvest mice. The greater the degree of *subsidence*, the greater the potential for *catastrophic* flooding of long duration. The 1983 flooding of the New Chicago Marsh in Alviso is an example of such potential flooding in a deeply subsided *marsh*. Coyote Creek overtopped, flooding all of Alviso, the New Chicago Marsh, and all the adjoining salt ponds. The *marsh* remained flooded for weeks, and *levee* tops surrounding the *marsh* (potential escape cover) were also underwater. Routine flooding and draining associated with conventional methods of waterfowl *marsh* management in Suisun Marsh also causes widespread, prolonged submergence of salt marsh harvest mouse habitat. Overtopping of *dikes* by storm *tides* is a common phenomenon in San Francisco Bay during extreme high *tides* that will probably increase with rising sea level, and may be exacerbated by increased storm intensity predicted by global warming (see section I.D.e.). Therefore, even *diked* salt marshes actively managed for long-term recovery of the salt marsh harvest mouse (Shellhammer 1989) may be at risk of *catastrophic* flooding.

One response of salt marsh harvest mice to flooding is movement to high ground, such as old *dikes* (Dixon 1908, Fisler 1965). During extreme flooding of the *marsh*, there is increased dispersal of salt marsh harvest mice from Mare Island strip marshes across Highway 37, which

can result in mortality from road kill (Wertz-Koerner 1997). Less extreme *tides* or floods that do not fully submerge *marsh* vegetation may not induce detectable dispersal (Hulst 2000). Movements across Hwy 37 are the exception. The more common threat to salt marsh harvest mice is that they are forced to the top of *Sarcocornia* as the highest high *tides* of the year rise and the animals are taken by predators. In marshes with a small total area of *Sarcocornia* it is surmised (Shellhammer pers. comm. 2005) that the death rate to predation and drowning exceeds the birth and immigration rate, and that these narrow marshes usually lose any salt marsh harvest mice.

Contaminants—The degree to which chemical contaminants, such as heavy metals, organochlorines, and PCBs (**Appendix E**) affect the quality of salt marsh harvest mouse habitat is not known. Initial studies in San Francisco Bay and San Pablo Bay that analyzed small mammal tissue samples for selected contaminants were inconclusive for salt marsh harvest mice (Clark *et al.* 1992). The presence of relatively high concentrations of contaminants (*e.g.*, mercury, lead, cadmium, selenium) at salt *marsh* sites with some of the largest or most dense *populations* of salt marsh harvest mice, such as Mare Island, Castro Creek Marsh, and Calaveras Point, suggests that contaminants may not be an overriding factor in habitat quality or reproductive success of this species.

Salt marsh harvest mouse habitat is at risk of contamination due to oil spills, particularly along major gas and oil pipelines alongside Highway 680.

B. *Cordylanthus maritimus* ssp. *maritimus* (salt marsh bird's-beak)

1) Brief Summary

Cordylanthus maritimus Benth. ssp. *maritimus* (salt marsh bird's-beak) of the south-central California coast was federally listed as endangered in 1978 (U.S. Fish and Wildlife Service 1978), and listed as endangered by the State of California in 1979 (California Department of Fish and Game 2005). It has been assigned a recovery priority number of 6, according to the 2007 Recovery Datacall for the Carlsbad field office of the U.S. Fish and Wildlife Service, based on its high degree of threat, low potential for recovery, and status as a subspecies (U.S. Fish and Wildlife Service 1983). A final recovery plan was prepared in 1985 (U.S. Fish and Wildlife Service 1985a). An isolated, and presumed *extirpated*, *population* at Morro Bay, San Luis Obispo County, California, was considered *Cordylanthus maritimus* ssp. *palustris* (Point Reyes bird's beak), at the time of listing, and thus was not treated in the 1985 recovery plan. Living *populations* rediscovered at Morro Bay in 1986, prompted taxonomic reinterpretation of the Morro Bay *population* which was subsequently classified as *C. m.* ssp. *maritimus* (Chuang and Heckard 1986). Because this *population* occurs with the endangered *Suaeda californica* (California sea-blite) in Morro Bay, it is included in this draft recovery plan. While adequate data are not available to assess long-term decline of *Cordylanthus maritimus* ssp. *maritimus* in Morro Bay, existing *populations* do face serious threats.

2) Description and Taxonomy

Description. *Cordylanthus maritimus* ssp. *maritimus* is an annual hemiparasitic plant in the Orobanchaceae (broom-rape family; **Figure II-13**). The popular name “bird’s-beak” refers to the curved, somewhat tubular flowers and bracts. The flowers of some *Cordylanthus maritimus* taxa have showy pale pink pouches with darker purple lips on purplish-green plants. Other taxa have pale grayish-green foliage and less conspicuous white flowers with dark brownish-purple lips. The flowers of *Cordylanthus maritimus* ssp. *maritimus* develop in loose to dense spikes 2 to 9 centimeters (0.8 to 3.5 inches) long. The hairiness of the foliage and stems is variable, and most plants have visible salt-encrusted glandular hairs. *Cordylanthus maritimus* may occur as short, erect, scarcely branched plants, or as plants with a profusion of spreading or ascending branches. The seeds are borne in capsules that mature from mid-summer through fall.



FIGURE II-13. *Cordylanthus maritimus* ssp. *maritimus* (Kendal Morris/USFWS)

Taxonomy. At the time *Cordylanthus maritimus* subsp. *maritimus* was listed, the genus *Cordylanthus* was placed in the Scrophulariaceae (figwort family). However, based on molecular systematic studies using DNA sequences of three *plastid genes*, Olmstead *et al.* (2001) transferred the *hemiparasitic* group Castillejiinae, including *Cordylanthus*, to the Orobanchaceae. This systematic treatment will be followed in the upcoming revision of the Jepson Manual.

Additional molecular *phylogenetic* analysis, initiated as part of the above cited studies, indicates that *Cordylanthus* is not a monophyletic genus (Tank and Olmstead 2008). In accordance with these findings Tank *et al.* (2009) recognize the genus *Chloropyron* and a previously published name *Chloropyron maritimum* (Nutt. ex Benth.) A. Heller subsp. *maritimum* for salt-marsh bird’s-beak. This combination will also be recognized in the

upcoming revision of the Jepson Manual. However, the *taxon* continues to be called *Cordylanthus maritimus* ssp. *maritimus* on the Federal List of Threatened and Endangered Wildlife and Plants (List) pursuant to the Endangered Species Act (Act) (16 U.S.C. 1531 et seq.), and this is the name that will be used in this recovery plan.

The species is divided into northern and southern coastal subspecies, and an inland subspecies. *Cordylanthus maritimus* ssp. *maritimus*, the southern California coastal subspecies, is distinguished from the northern ssp. *palustris*, mainly by geographic distribution in that it occurs from Morro Bay south through southern California. It is also distinguished by branching patterns, growth habit, narrower and more acute leaves, and variations in seed size and floral traits (Chuang and Heckard 1973, 1993). Though the *population* of *Cordylanthus maritimus* ssp. *maritimus* at Morro Bay is addressed in this draft recovery plan for reasons stated above, all other *populations* of the subspecies are addressed in the *Salt Marsh Bird's-Beak Recovery Plan* (U.S. Fish and Wildlife Service 1985a). The Morro Bay plants were not considered *Cordylanthus maritimus* ssp. *maritimus* when the 1985 Salt Marsh Bird's-Beak Recovery Plan was written.

The three intergrading subspecies have distinct ecological and geographical distributions. *Cordylanthus maritimus* ssp. *canescens* (hoary salt marsh bird's-beak) is a widely distributed, but uncommon, plant of inland saline/alkaline wetlands of the Great Basin; *Cordylanthus maritimus* ssp. *maritimus* (salt marsh bird's-beak), an endangered *tidal marsh* plant limited to few *populations* in southern California and Baja California, Mexico; and *Cordylanthus maritimus* ssp. *palustris* (Point Reyes bird's-beak), a similar rare *tidal marsh* plant from San Francisco Bay to Oregon.

Although Chuang and Heckard (1973) concluded that the morphological differences are sufficient to warrant taxonomic distinction below the species rank, they noted that specimens morphologically intermediate between *Cordylanthus maritimus* ssp. *canescens* and ssp. *maritimus* occur in saline inland soils of southern California near the coast. Chuang and Heckard (1973) further observed that specimens of putative ssp. *maritimus* at the northern end of its range (south of Morro Bay) resemble ssp. *palustris* (Chuang and Heckard 1973), and Chuang and Heckard (1986) later reclassified the Morro Bay *population* from ssp. *palustris* to ssp. *maritimus*, realigning the subspecies range limits. Chuang and Heckard (1973) cautioned that ssp. *maritimus* is a variable group, and is itself intermediate between ssp. *canescens* and ssp. *palustris*.

3) Population Trends and Distribution

Historical Distribution. Historically, *Cordylanthus maritimus* ssp. *maritimus* was widespread near the upper edges of coastal salt marshes from Morro Bay in San Luis Obispo County to San Diego County and northern Baja California. Presently, it occurs only in scattered sites at fewer than 10 remnant salt marshes. Half of the original occurrences are now considered *extirpated*.

Current Distribution. The Morro Bay *population* of *Cordylanthus maritimus* ssp. *maritimus* was not detected between 1912 and at least 1970 (Hoover 1970, Chuang and Heckard 1973), but has been reported since the 1980s (L. Heckard pers. comm. 1986). The Morro Bay *population*

consists mostly of small subpopulations (hundreds to thousands of plants) in very localized fluctuating colonies in two major local populations (P. Baye pers. obs. 1998-1999; **Figure II-14**). One is found between Sweet Springs Nature Preserve and Shark's Inlet, the second is southeast of the terminal widening of the sandspit (M. Walgren *in litt.* 2006). In the 1990s there was a population southwest of Cuesta Inlet near a public access point where the habitat is subject to trampling. This population has since disappeared; the exact cause of the disappearance is unknown (M. Walgren *in litt.* 2006). In 2004, the population near Sweet Springs Nature Preserve consisted of 2 subpopulations, with more found in some years. The sandspit supports no subpopulations in some years. It is unclear if the population is in decline or simply has variable success from year to year (M. Walgren *in litt.* 2006). In 2004, the salt marsh bird's-beak population at Morro Bay totaled roughly 1,300 plants, about 1,000 of which were in the sandspit locality (M. Walgren *in litt.* 2006).

Populations of *Cordylanthus maritimus* typically fluctuate by orders of magnitude among years (Parsons and Zedler 1997). Population fluctuations in *Cordylanthus maritimus* ssp. *palustris* may relate to rainfall and vegetation structure, but the relationship is neither simple nor well understood. High rainfall appears to correspond with large population size in ssp. *maritimus* in more arid southern California (Parsons and Zedler 1997, B. Grewell pers. comm. 2000).

4) Life History/Ecology

Reproduction. Factors considered important to the reproductive status of *Cordylanthus maritimus* ssp. *maritimus* include the small numbers of individuals, the isolation of individual plants, pollination, herbivory, seed production, seed dispersal, seed dormancy, seed germination, and seedling habitat.

Cordylanthus species were once thought to be self-incompatible (Chuang and Heckard 1973); however, later work by Parsons and Zedler (1997) indicates that there is some degree of self-compatibility and that both cross- and self-pollination may increase with flower manipulation by insect visitors. Whether self-pollination alters the viability of the seeds needs further investigation.

Specific pollinators of *Cordylanthus maritimus* ssp. *maritimus* at Morro Bay are unknown (M. Walgren *in litt.* 2006). Bees are thought to be the principal pollinators of ssp. *maritimus* at other locations (Parsons and Zedler 1997). The flower structure suggests that only bees would be effective pollinators; bumblebees (*Bombus* spp.) may be the most efficient and effective (Proctor *et al.* 1996, Faegri and van der Pijl 1979). Small native halictine bees have been observed visiting ssp. *maritimus* flowers at Ormond Beach, Ventura County. At Point Mugu, Ventura County, four species of bees and two species of flies appear to pollinate the flowers (U.S. Fish and Wildlife Service 1985a). Upland habitats near tidal marsh occupied by *Cordylanthus maritimus* are likely needed to support pollinating insects that do not nest in tidal marsh, such as most bees (Callaway and Zedler 2004).

The flowering period for *Cordylanthus maritimus* ssp. *maritimus* is May to October. Seed output averages between 15 to 20 seeds per capsule (Chuang and Heckard 1973). Many factors may reduce seed set. Pre-dispersal seed predation in *Cordylanthus maritimus* ssp. *maritimus* can be

caused by *lepidopteran* larvae and locusts (U.S. Fish and Wildlife Service 1985a). The salt marsh snout moth, *Liphographus fenestrella* (Parsons and Zedler 1997) and leaf roller moth larvae (*Platynota stultana*; U.S. Fish and Wildlife Service 1985a) are known seed predators. Nothing is known of post-dispersal seed predation in *Cordylanthus maritimus*.

Unlike *perennial* plants, the annual *population* of *Cordylanthus maritimus* depends entirely on yearly seed germination and seedling establishment. Physical factors such as currents, *tides*, wave action, and sheet erosion are among the ways seeds are moved around within and between marshes. The seeds of *Cordylanthus maritimus* ssp. *maritimus* have a honeycombed surface that traps air bubbles and makes them highly buoyant. They have been shown to float for up to 50 days and floatation may be the primary local dispersal mechanism for *Cordylanthus maritimus* ssp. *maritimus* (Newman 1981). Animals, especially birds, may carry the seeds on their feet, or in their fur, feathers, or digestive systems (U.S. Fish and Wildlife Service 1985a).

Cordylanthus maritimus persists through unfavorable years as a dormant *seed bank* (Parsons and Zedler 1997) because high densities and abundance of standing plants may follow years of extremely low seed production. The longevity of the *marsh* soil *seed bank* of this species is not known, but artificially stored seed of ssp. *maritimus* have remained *viable* for over 11 years (Parsons and Zedler 1997). Dry storage of seeds for two years enhanced germination by 230 percent over germination of fresh seeds (U.S. Fish and Wildlife Service 1985a). Germination also increased with scarification (cutting the seed coat using abrasion, thermal stress, or chemicals) or vernalization (the acquisition of the competence to flower in the spring by exposure to the prolonged cold of winter). (Newman 1981).

Availability and abundance of seedling habitat may be an important factor limiting reproduction in ssp. *maritimus*. The range of *salinity* associated with growth of ssp. *maritimus* is 5 to 33 parts per thousand, but pulses of *freshwater* from flooding or rainfall are probably necessary for germination (Parsons and Zedler 1997). *Salinity* at the time of germination usually cannot exceed 12 parts per thousand (Newman 1981).

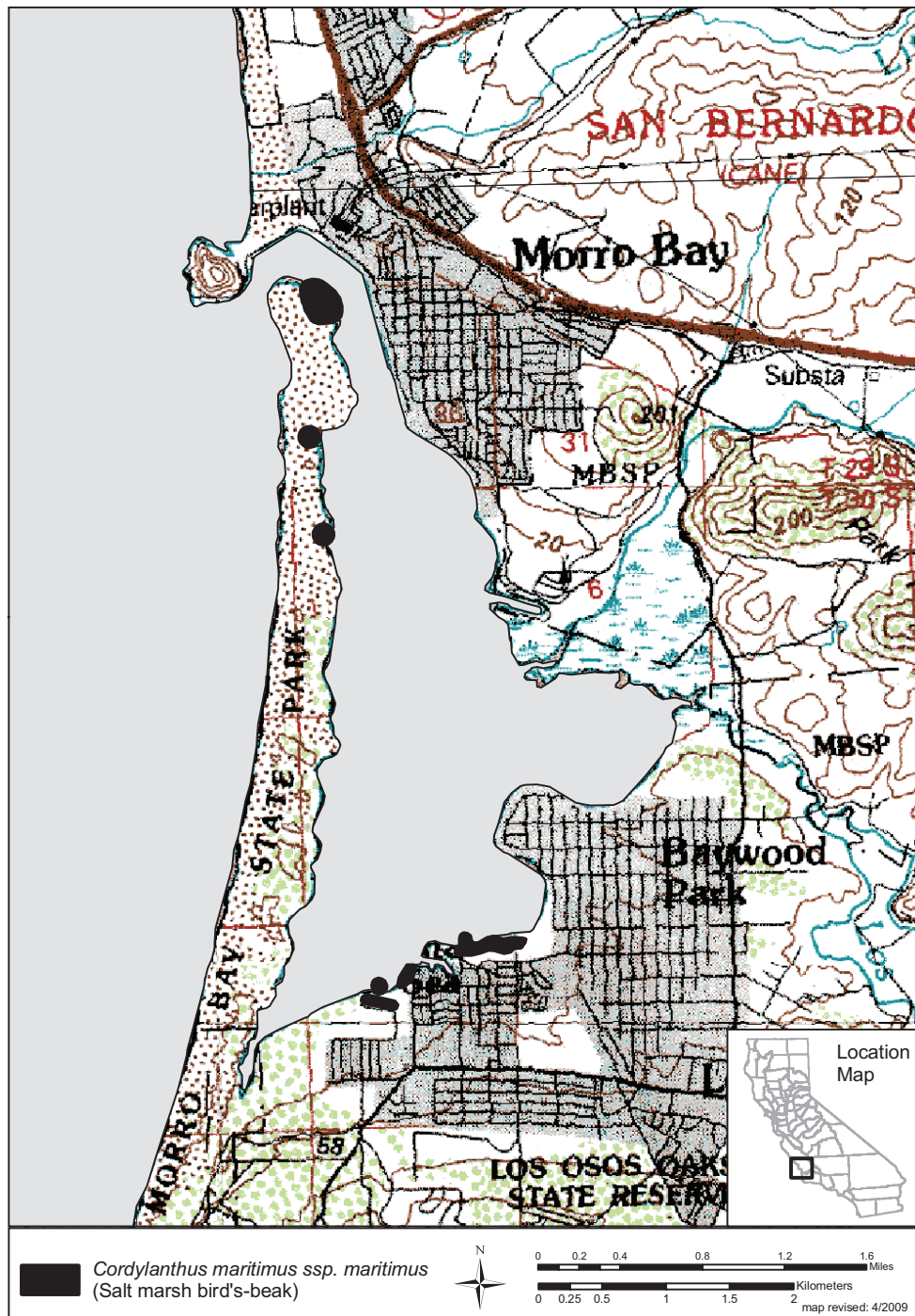


Figure II-14.

Figure II-14. Distribution of *Cordylanthus maritimus* ssp. *maritimus*

5) *Habitat Characteristics/Ecosystem*

Cordylanthus maritimus ssp. *maritimus* occurs in variable habitats throughout its range. It appears to favor the middle to high *marsh* zone, but may range toward upper and lower extremes in some cases. In Morro Bay it occupies a narrow margin of *estuary* edge at the high *tide* line (M. Walgren *in litt.* 2006), with one *population* found in *brackish* to salt *marsh* and one in relatively recently formed salt *marsh* at the north end of the sandspit (P. Baye pers. obs. 1998-1999). *Populations* generally occur in areas with low *salinity* in the spring and low vegetative cover (Newman 1981, Dunn 1981). *Cordylanthus maritimus* ssp. *maritimus* is found on sandy *marsh* substrates with relatively sparse, short salt *marsh* vegetation, and is usually absent or declining in dense, tall salt *marsh* vegetation (Newman 1981, Kelly and Fletcher 1984, Parsons and Zedler 1997). Dense vegetation may inhibit growth due to shading or reduced water availability. *Cordylanthus maritimus* ssp. *maritimus* may increase in abundance in response to disturbances that reduce vegetation cover (Vanderweir and Newman 1984, Parsons and Zedler 1997). However, it is vulnerable to crushing and trampling, as it is easily broken off and tends to occur at higher, less muddy *marsh* elevations where foot traffic is concentrated (Zedler 1982, Zedler 1984, M. Walgren *in litt.* 2006).

In Morro Bay, *Cordylanthus* is found in typical *estuary* edge vegetation (M. Walgren *in litt.* 2006). It is associated with *Sarcocornia pacifica* (pickleweed), *Distichlis spicata* (salt grass), *Frankenia salina* (alkali-heath), *Limonium californicum* (sea-lavender) and occasionally *Cuscuta salina* (saltmarsh dodder; M. Walgren *in litt.* 2006).

Members of the genus *Cordylanthus* are *hemiparasitic*. Their roots form haustoria to obtain water and nutrients through the roots of other host plants. It is not known to what degree individuals of this subspecies are dependent on their hosts. *Cordylanthus maritimus* can grow without host plants (Chuang and Heckard 1971), but hemiparasitism may permit them to flourish in the hot, dry, higher soil-*salinity* conditions of summer (Vanderwier and Newman 1984). Under experimental conditions, *Cordylanthus* exhibits variation in biomass depending on host species (Fink and Zedler 1990). The host plants of the *Cordylanthus maritimus* ssp. *maritimus* are unknown (M. Walgren *in litt.* 2006).

Dense vegetation may inhibit growth of *Cordylanthus maritimus* ssp. *maritimus* due to shading or reduced water availability. For a period of time following germination, seedlings live independently. Experimental work on *Cordylanthus maritimus* ssp. *maritimus* indicates that soil nitrogen limits reproductive capacity of individual plants (Parsons and Zedler 1997). The largest *populations* of ssp. *palustris*, however, are on sandy *marsh* substrates (Russell 1973) with sparse and low vegetation cover, suggesting that unproductive environments, rather than productive nitrogen-rich environments, favor abundance in the field.

6) *Critical Habitat*

Critical habitat has not been designated for this subspecies.

7) *Reasons for Decline and Threats to Survival*

Most species covered in this draft recovery plan are threatened by similar factors because they occupy the same *tidal marsh* ecosystem. These general threats, faced by all covered species, are discussed in greater detail in the Introduction section of this draft recovery plan (section I.D.). Specific threats to *Cordylanthus maritimus* ssp. *maritimus* are described below.

All the *Cordylanthus maritimus* ssp. *maritimus* known localities, as well as potential habitat around Morro Bay, are at risk from impacts of *non-native* plants, including *Carpobrotus edulis* (iceplant) and trees and shrubs such as (*Eucalyptus globulus* (eucalyptus), *Myoporum laetum* (myoporum), and *Cupressus macrocarpa* (Monterey cypress). *Non-native* plants may crowd out the subspecies or degrade its habitat through shading, litter fall, or *freshwater* drawdown.

The largest *subpopulation* of *Cordylanthus maritimus* ssp. *maritimus* at Morro Bay is on the bay side of the barrier sand spit, close to a large expanse of mobile sand and at risk of being buried by a high wind event (P. Baye pers. observ. 1997-2000). While dune movement is a natural process, this area has been affected by past deposits of dredged sand. The low numbers and small area currently occupied by the subspecies at Morro Bay increase the risk of a substantial portion of the local *population* being destroyed by a natural process involving an unnatural resource (dredged sand).

Two of the localities of the subspecies at Morro Bay occur in marshes adjacent to residential locations in the Los Osos area. These habitats are narrow, sandy high salt *marsh* fringes, potentially subject to disturbance impacts such as trampling by humans, horseback riding, and boat haul-outs. Currently, dozens of small boats are routinely left hauled out on the high *tidal marsh* zone in several areas around the bay. Although recent disturbance levels have been low in many areas, continued increase in residential *population* and recreational pressures may adversely affect this subspecies. Demand for flood control or shoreline stabilization near residential areas—particularly in light of rising sea level and higher extremes of storm and wave energy—could exert pressure to harden shorelines or build *berms* in habitat areas.

While there has been some loss of potential habitat for *Cordylanthus maritimus* ssp. *maritimus* in areas of shoreline hardening, such as around developed portions of the City of Morro Bay, the great majority of this occurred prior to the recognition of the Morro Bay plants as ssp. *maritimus*. Since at that time, the plants were thought to be the more common *Cordylanthus maritimus* ssp. *palustris*, impacts to the plants were not assessed with the same degree of scrutiny as impacts to the more rare subspecies would have been; therefore, the degree of impact is not known.. Ongoing development for housing and other purposes in upland habitats near *tidal marsh* is likely to reduce native pollinators of the subspecies.

Nearly all the threats faced by *Cordylanthus maritimus* ssp. *maritimus* at Morro Bay are heightened by the low numbers and small area of distribution of the subspecies there. Small *populations* have increased vulnerability to extinction due to *catastrophic* events like severe droughts, storms, fires, pollution spills, *non-native* species invasion, or epidemics (Schonewald-Cox *et al.* 1983). Another factor is natural variability in birth and death rates: a chance cluster of years of high death rates or low birth rates is likely to result in the extirpation of small

populations. At low *population* sizes, *genetic* and evolutionary effects become important, including loss of *genetic* diversity due to *founder* effects, *genetic* drift, *inbreeding*, and *inbreeding* depression. In December, 2003, Morro Bay experienced an earthquake (centered near Paso Robles) that uplifted portions of potential habitat. The uplift was patchy but in some areas amounted to a foot or more, enough to make formerly suitable habitat uninhabitable by the subspecies.